THE LIFECYCLE COST OF RENEWABLE ENERGY TECHNOLOGIES WITH A FOCUS ON END-OF-LIFE AND RARE-EARTH ELEMENTS

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MFM 2018-2019

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PROJECT SUBMITTED IN FULFILLMENT OF THE DEGREE OF MASTERS IN FINANCIAL MANAGEMENT

Executive summary

In the first part of this report, a life-cycle assessment of renewable energy technologies is conducted, with a focus on the end-of-life stage. In the second part, rare-earth elements and other critical elements are investigated to assess their impact on renewable energy.

In the first chapter, the levelized cost of energy is found to be of 68 €/MWh for onshore wind energy, 88 €/MWh for offshore wind and 59 €/MWh for utility scale photovoltaics. In comparison with non-renewable energy, combined-cycle gas turbines have a levelized cost of energy of 48 €/MWh wen operating close to optimal capacity. However, combined-cycle gas turbines have significantly higher carbon emissions and, when adding the social cost carbon emissions, the levelized cost of energy increases to 73 €/MWh. As a result, onshore wind energy and utility scale photovoltaics could be considered as cheaper, when taking the cost to society into account. In practice, combined-cycle gas turbines operate at a lower capacity since they are used as a back-up energy source for times at which the supply of other energy sources is lower than demand. When using the actual capacity in Germany, the levelized cost of energy increases to 71 €/MWh and 95 €/MWh when the social cost of carbon is included.

For onshore wind, offshore wind and solar energy, the levelized cost of energy is split up in 3 stages. The installed costs are incurred at the beginning of the project, the operating and maintenance expenses are incurred from year 1 until the end of the lifetime, and finally there are the end-of-life costs. For each of these 3 technologies, the installed cost makes up the biggest part, ranging between 72% and 76%. Of the other costs, almost all costs are made up out of operating and maintenance expenses. End-of-life costs are negligible for onshore wind energy, since almost all decommissioning costs are offset by the salvage value of scrap materials. For offshore wind-energy, the decommissioning costs are substantial, at 412.060 €/MW in the base case, after subtracting salvage value. These decommissioning costs make up 3,34% of the total LCOE for offshore wind energy. For solar energy, the manufacturers of solar panels are legally obliged to bear the end-of-life costs in Europe. Because of this, the end-of-life costs are included in the installed costs.

Most materials in wind turbines are recycled. For wind turbines, these include mainly steel and copper. The bottleneck in recycling for wind turbines are the blades, which are made mainly of fiberglass reinforced plastic. At the moment, most turbine blades are landfilled or incinerated. Reusing the entire blade is the best option, but this is limited to smaller blades because of transport issues. Recycling currently leads to a reduction of the quality of the material, but technology and legislation improvements might result in increasing recycling rates. Some wind turbines use permanent magnets, which include rare-earth elements. These could also be recycled without significant reductions in quality, but with substantial losses of iron in the process.

For solar energy, recycling rates are also high. Silicon panels and thin-film panels are, respectively, made for 90% and 95% of glass, polymer an aluminium. Together, these 2 technologies make up 95% of installed solar capacity worldwide. There are established recycling industries for glass, aluminium and polymers and they can be easily recycled and reused. However, silicon panels contain small amounts of silicon, silver and some other elements that present recycling difficulties. Thin-film panels also contains small amounts of copper, zinc and trace amounts of other elements that are not recycled, and some are potentially hazardous. Some companies (First Solar) have dedicated recycling facilities that recover most of the valuable materials from panels and in France, a dedicated solar recycling facility is opened that can recycle all of the components, which is not possible with traditional recycling plants.

The carbon footprint of wind energy is the lowest, at around 10 gCO_2/kWh, followed by solar energy at around 48 gCO_2/kWh. This is very low. For example, combined cycle gas turbines have 490 gCO_2/kWh and it is around 820 gCO_2/kWh for coal.

For batteries, there is no a general method to calculate the levelized cost. We found that economic viability of battery storage is highly case dependent and varies according to the technology used, the region and revenue streams. Revenue streams include energy arbitrage and bill management. We found that batteries are economically viable in Germany and California for residential PV+storage applications, however, it is dependent on government subsidies and incentives. Battery applications for utility scale PV+storage, wholesale, commercial and transmission and distribution are economically viable but highly dependent on different revenue streams (such as energy arbitrage, bill management etc.) for geographies the study conducted. Additionally, cost of batteries is expected to decrease further, specifically for the case of li-ion batteries in the future, of which the price is expected to decline 28% by 2022 with a CAGR of 8%.

In the second chapter, we found that 71% of rare-earth elements originated from China in 2018. This is already substantially lower compared to 2010, when 95% came from there. Other countries were concerned when China reduced its exports with 40% in 2010, so they increased rare-earth element production. Other big producers are Australia and the United states, with 12% and 9% of production respectively. The production in China of 71% of the world production is still compared to China's reserves, which make up 37% of the world reserves of rare-earth elements. The total reserves in the world are over 700 times larger than the world's rare-earth element production in 2018.

In China, rare-earth mining and processing industry has undergone significant consolidation, driven by the Chinese government. Chinese rare-earth production is now dominated by 6 companies: the big six. Outside China, production is dominated by Lynas in Australia and MP materials In the United States.

Wind turbines could contain over 200kg of rare-earth elements per MW. This is because some turbines contain permanent magnets, which contain neodymium, dysprosium and praseodymium. These are called NdFeB magnets. These magnets were originally invented by a US company and a Japanese company. Today, the Japanese company, Hitachi, has over 600 patents over these magnets. However, the production is also dominated by China. It is estimated that over 80% of permanent magnets are manufactured in China and, of the 13 Hitachi licensees, 8 are located in China.

At the moment, most wind turbines don't contain rare-earth elements and use a different technology. However, as wind turbines become bigger and more wind turbines are located offshore, the use of rare-earth elements is expected to increase. When the amount of rare-earth elements used increases, there could be shortages in supply. Studies project that demand for neodymium in 2030 could be 7 times higher than the supply of 2017, and this is only the demand from wind turbines. In 2010, only 1% of neodymium produced was used for wind turbines.

While it is clear that there are enough reserves, concerns exist about the ability to increase production fast enough. Opening new mines could take over 10 years and the supply by recycling of wind turbines lags behind over 20 years. However, as already mentioned, there is a trend to open mines outside China, and some could already be opened as of 2020. Even in case of shortages of supply, there are sufficient substitutes. Most turbines today don't use permanent magnets but use doubly-fed induction generators which don't contain rare-earth elements. Another option in the future is superconducting wind turbines, which also don't use rare-earth elements. Furthermore, there is the option to reduce rare-earth element content, for example by using hybrid wind turbines.

We estimate that the cost of rare-earth elements in the total levelized cost of energy is less than 1% for onshore wind and even lower for offshore wind. However, if the prices of rare-earths increases to the peak prices in 2011, they could make up over 5% of the levelized cost of energy.

The carbon footprint of rare-earth elements used in wind turbines is 0,69 gCO_2/kWh, which is around 7% of the total carbon footprint of onshore wind and 0,084% of the carbon footprint of coal. However, on a normalized scale, human toxicity, aquatic ecotoxicity, eutrophication of fresh water and particulate matter are found to have a bigger environmental impact. Rare-earth mining in China appears to be much more environmental damaging than elsewhere, as it is estimated that the environmental impact of a rare-earth mine would be 60-80% lower in Europe compared to China.

We looked at critical elements for solar PV at element level and technology level using the weighted average of different risk criteria and indicators. Thin-film technologies are chosen because of the overall criticality of the elements used. Technologies considered are CdTe and CIGS panels. As a result, we found that, on element level, indium is the element with the highest risk, while copper has the lowest risk. On technology level, CdTe panels have consistently lower supply chain risk that CIGS panels using different weighting methods. Additionally, in order to implement Paris agreement, production of some elements need to rise severalfold by 2050. For example, production of indium needs to rise more than 12 times and this figure is more than 7 times for neodymium.

We identified cobalt, lithium, manganese and graphite as critical elements for batteries. For these elements, there is supply risk to the EU, particularly due high concentration risk. For example, 64% of cobalt comes from the Democratic Republic of Congo and 69 % of graphite comes from China. It is expected that cobalt and lithium demand will be 3 and 3.5 times higher respectively by 2025 for rechargeable batteries. Lithium reserves itself worldwide are enough to meet the worldwide demand in the coming decades, but there currently are only few high-grade lithium processors for high grade lithium for batteries.

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List of acronyms

BEIS	Department for Business, Energy & Industrial Strategy
CAGR	Cumulative aggregate growth rate
CAPEX	Capital Expenditures
CCGT	Combined cycle gas turbine
DD	Direct drive
DFIG	Doubly-fed induction generator
DOE	United States Department of Energy
GHI	Global Horizontal Irradiance
HREE	Heavy rare-earth element
IEA	International Energy Agency
IRENA	International Renewable Energy Agency
ISE	Institute for Solar Energy Systems
kW	Kilowatt
kWh	Kilowatt hour
LAMP	Lynas Advance Materials Plant
LCOE	Levelized cost of energy
LCOES	Levelized cost of energy storage
LREE	Light rare-earth element
MW	Megawatt
MWh	Megawatt hour
NdFeB	Neodymium Iron Bohr
NREL	National Renewable Energy Laboratory
NUF	Neutralization Underflow
O&M	Operating and Maintenance costs
PMSG	Permanent magnet synchronous generator
PV	Photovoltaic
PVOUT	Photovoltaic Electricity Output
REE	Rare-earth element
WACC	Weighted-average cost of capital
WEEE	Waste electrical and electronic equipment
WLP	Water leached Purification Residue

I. Research question

Renewable energy is crucial to combat climate change and to reduce reliance on fossil fuels. These renewable energy sources are considered green because of they are considered to have a low carbon footprint. However, there are some factors which might cause them to be less green than expected.

Renewable energy technologies like wind turbines and solar panels have only been developed during the last decades on a large scale. Given that the lifetimes are often more than 20 year, the end-of-life costs might seem like an issue for far into the future. However, more and more solar and wind projects are currently reaching the end-of-life stage. This could create a lot financial and non-financial costs. This leads to the first research question: *"What are the end-of-life costs for renewable energy technologies and what is the share in the total life-cycle cost?"*

Another issue with renewable energy is that they tend to use rare-earth elements or other critical raw materials. In 2010, 95% of rare-earth elements originated from china and concerns exist about the impact of rare-earth mining and processing on the environment and the people working and living in and around the mines. Moreover, an increase in the use of renewable energy technologies could mean disruptive demand for rare-earth elements. If the supply could not be increased fast enough, this could hinder the energy transition. This concern does not only exist for rare-earth elements, but could be extended to other critical raw materials, which could be rare or mined in countries with low environmental standards. This results in our second research question: *"How do rare-earth elements and other critical raw materials affect the sustainability and future of renewable energy technologies?"*

II. Methodology

Given the wide scope of our research, we focused on secondary data. Substantial amounts of research have been done regarding the topic, but the research is scattered through multiple studies. Our work mainly consisted of synthesising existing academic research in order to consult the company in the most efficient way. Sometimes, we used the data of other research and executed our own analyses. Next to secondary data, we also conducted an interview with an engineer who works at the company. This was mainly useful for understanding the technological aspects behind energy. The first research question is answered in the chapter 1 and the second research question is answered in chapter 2.

A drawback of renewable energy is the high variability in supply. Because of this, the cost of batteries is also investigated.

1. Lifecycle cost

In order to answer the first research question, we first provided an overview is provided of the installed capacity of each renewable energy source. Based on this, the most important ones were selected: wind energy and solar energy. While hydropower is the biggest one, it is not included because it is not relevant for Belgium.

To calculate the share of end-of-life costs in the total life-cycle cost, a life-cycle assessment is most appropriate. For this, we needed to link all costs of a certain product to each stage in the lifetime. Therefore, we mainly investigated academic research and renewable energy reports. These include reports from companies (e.g. Lazard) or government departments (e.g. US Department of Energy). The data from these reports was used for the calculation of the cost of energy.

1.1. Levelized cost of energy

The levelized cost of energy (LCOE) methodology was used to obtain the life-cycle cost. Aldersey-Williams & Rubert (2019) identify 2 different, but similar, approaches to compute the LCOE: the approach of the department for Business, Energy & Industrial Strategy in the UK (BEIS) and the approach of the National Renewable Energy Laboratory. They note that under certain simplifying assumptions, both methodologies should return the same value. We used the BEIS-approach.

Levelized cost of energy calculation

$$LCOE_{BEIS} = \frac{NPV_{Costs}}{NPE} = \sum\nolimits_{t=1}^{n} \frac{C_t + O_t + V_t}{(1+d)^t} \Big/ \sum\nolimits_{t=1}^{n} \frac{E_t}{(1+d)^t}$$

Equation 1: LCOE formula using the BEIS-approach (Aldersey-Williams & Rubert (2019)

The BEIS-approach calculates the net present value (NPV) of all costs over the lifetime of an asset and divides this by the net present energy (NPE). Equation 1 shows how this is calculated, where t is the period ranging from year 1 to year n, C_t the capital costs during period t, O_t the fixed operating costs during period t, V_t the variable operating costs during period t, E_t the energy production during period t, d is the discount rate and n is the lifetime of the project in years. As a result, the costs per unit of energy (e.g. 1 MW) is found.

The NREL-approach is similar to the BEIS-approach, but slightly different. The main difference is that NREL-approach uses annuity-based recovery factor for initial capital expenditures. This way, all costs are spread evenly over the years, and there’s no need to discount. As a result, the cost per year can be divided by the energy production per year (Aldersey-Williams & Rubert, 2019).

Advantages and disadvantages

The LCOE-methodology is used because it is widely adopted, and it takes all costs during the lifetime into account. Furthermore, since it computes the cost per MWh, it is easy to compare the cost of different projects. The LCOE could also be interpreted as the energy price required for the investors to have a fair return, equal to the discount rate used (Albrecht & Laleman, 2015). It considers all costs during the lifetime of a project, and thus derives the lifetime cost of energy. Furthermore, Aldersey-Williams & Rubert (2019) provide a theoretical justification for the correctness of the model.

There are also several drawbacks of using the LCOE methodology. These include the dependence on discount rate, inflation effects, and sensitivity to uncertainty about future commodity costs (Aldersey-Williams & Rubert, 2019). Also, it does not consider variation in energy prices throughout the day or de difference in value between dispatchable and intermittent generation (Jaskow, 2011).

1.1.1. General methodology

To improve comparability, similar assumptions and methodology were used for wind energy and solar energy. We also calculated the LCOE ourselves because this made it possible to look at the share of each stage in the LCOE, including end-of life costs. Furthermore, it allowed us to perform our own sensitivity analyses etc. We also compared it to the results of other studies.

The installed cost, operating and maintenance expenses (O&M), end-of-life costs, capacity factors or irradiance levels, the lifetime of the plant and discount are the input parameters for the calculation. The installed cost includes all costs which are incurred before the plant is operational. O&M include the costs of operating and maintenance, so the costs incurred from the moment the plant is operational, until the decommissioning stage. End-of-life costs are the costs of decommissioning, offset by the salvage value of the materials. Capacity factors and irradiance levels are technology-specific and are explained under the specific section. Most LCOE-studies often include these input parameters, except for end-of-life costs. As a result, most of our input parameters are based on these LCOE studies. This was either done by calculating averages across studies or adopting the same value as the study which best approximates Belgium. Only for the discount rate and end-of-life costs, a different approach is implemented.

Discount rate

As the LCOE-formula shows, the costs and produced energy need to be discounted. We used the same discount rate for each renewable energy technology. The advantage of this standardised discount rate is that it makes it easier to compare across technologies and LCOE differences won't be caused by differences in financing. Nonetheless, this assumption doesn't hold in reality, as there are differences in project-specific risk, as well as debt/equity ratio's across projects. This approach of using the same discount rate for each technology is also adopted by other sources, such as IRENA (2018) and Lazard (2018).

In line with the BEIS-approach, a hurdle rate was used, which is the minimum rate of return required for the project, based the pre-tax real basis. As a result, the pre-tax real weighted-average cost of capital (WACC) was used. According to Aldersey-Williams & Rubert (2019), both a real discount rate and a nominal discount rate are possible. If a real discount rate is used, it is important to be consistent and project all costs in real terms. Correspondingly, the LCOE found could be interpreted as the real energy price required throughout the lifetime to have an internal rate of return equal to the discount rate. If a nominal discount rate would be used in combination with nominal cost projections, the calculation would result in a LCOE equal to the nominal energy price required for the IRR to equal the nominal discount rate. Furthermore, we chose the pre-tax WACC, because we want to compute the LCOE without considering taxes or government subsidies.

Cost of equity	10,62%
Risk-free rate	1,23%
Unlevered beta	0,65
Levered beta	1,38
Market risk premium	6,80%
Cost of Debt	4,00%
Equity	40%
Debt	60%
Nominal WACC pre-tax	6,65%
Inflation	2,00%
Real WACC pre-tax	4,56%

Table 1: WACC calculation

Table 1 shows the WACC calculation. To compute the cost of equity, Damodaran's unlevered beta for the green and renewable energy industry is used (Damodaran, 2019). Based on this unlevered beta and a debt ratio of 60%, the levered beta is 1,38. The market risk premium is based on Damodaran's market risk premium for Belgium (Damodaran, 2019). Using the capital asset pricing model, the cost of equity of 10,62% was computed. The cost of debt is based on the average of the cost of debt for renewable energy technologies used by Fraunhofer ISE (Kost, Shivenes, Jülch, Nguyen, & Schlegl, 2018), who compute the cost of debt for Germany. The cost of debt in Belgium is assumed to be similar to Germany, so we used the same cost of debt. The inflation rate of 2% is based on the ECB's goal to keep inflation below, but close to 2%. Based on these input variables, a real pre-tax WACC of 4,56% was found.

For the calculation of the LCOE of combined cycle gas turbines, a different discount rate was used. For this, we changed the beta to a new beta. Instead of the beta for the green and renewable energy industry, we used the unlevered beta for the coal & related energy industry (Damodaran, 2019). This way, the unlevered beta increases from 0,65 to 0,78, resulting in an increase of the discount rate to 5,29%.

1.1.2. Specific methodology aspects for wind energy

As already mentioned, end-of-life costs are most often not mentioned or included in studies calculating the LCOE. For onshore wind turbines, the average was taken from different decommissioning estimates, which are legally required in some countries. Less studies exist for offshore wind turbines, so the estimate is based on a highly detailed and recent study for a project in Canada.

Another input parameter is the energy produced over the lifetime. This was computed using the nameplate capacity and the capacity factor. The capacity factor could be computed as the actual produced energy divided by the maximum amount of energy that could be theoretically produced. For example, a wind turbine with a nameplate capacity of 1 MW, could theoretically produce 8760 MWh[1] in 1 year. However, in practice, this never happens because the wind doesn't blow at the optimal speed all the time. As a result, the actual production will be much lower. The capacity factor depends on the

[1] There are 8760 hours in 1 year

location of the wind turbine, as well as on the wind turbine model. For example, a wind turbine with a nameplate capacity of 1MW and a capacity factor of 25%, is able to produce 25% of 8760 MWh.

For the lifetime of a wind farm, most LCOE-studies assume either a lifetime of 20 years or 25 years. We assumed a lifetime of 22 years and the decommissioning is assumed to happen in year 23. In order to compute the share of end-of-life costs in the total LCOE, the discounted decommissioning costs divided by the net present energy could be compared to the total LCOE.

This approach is used for both onshore and offshore wind turbines. Sensitivity analyses were also performed and added to the appendices.

1.1.3. Specific methodology aspects for solar energy

For solar energy, we considered end-of-life costs to be zero for because in Europe, producers, regardless of origin, are responsible for recycling of PV panels at the end of their life both financially and physically. However, it is possible that these costs are already reflected at the price of PV panels and effectively considered in the capital costs of PV panels if there are any. For our calculation of LCOE of PV panels, we took Fraunhofer institute LCOE calculation for Germany as a base case and adjusted it to the Belgian conditions.

An important consideration is irradiance levels, which determines how much solar energy hits the earth surface per m² annually. We found out that average irradiance level for Belgium is 1.125 kWh/m² per year and that translates 1.105 kWh/kW of PV output (PVOUT).

We chose an average lifetime of 25 years because it is used in most of the studies as lifetime for PV panels and most producers guarantee their systems for 25 years. However, it must be mentioned that the useful life of PV panels is expected to increase and, as a result, LCOE of PV panels are expected to decrease in the future.

1.2. Non-financial costs

The LCOE does not include non-financial costs. These are costs which need to be borne by society, for example waste and environmental damage. Because of this, the recycling practices and carbon emissions were also included. For this, we mainly investigated and synthesised existing literature.

The carbon emissions are typically calculated as gCO_2/kWh, which is the weight in gram of CO_2 per kWh. This is also the way we reported this. We also investigated academic research, which quantifies the cost of carbon emissions to society. This cost is quantified as the price per tonne of carbon emissions. Based on carbon emissions per kWh (gCO_2/kWh), the social cost of carbon per MWh could be computed. This was added to the total LCOE in the end, when the results were compared to combined cycle gas turbines. Because we added the cost of carbon to the LCOE of combined cycle gas turbines, it was important that we also excluded the cost of CO_2 certificates from the input parameters.

2. Rare-earth elements and other critical elements

To answer the second research question, we needed to investigate how rare-earth elements impact the future and sustainability of renewable energy technologies. For this, we first looked for general information about rare-earth elements (REE) in order to obtain a better understanding about their importance in today's world. This includes understanding the mining and refinement process of REEs, their use in modern technology etc.

In order to investigate how REEs impact the future of renewable energy technologies, we also investigated the amount and locations of the reserves. This was then compared to where REEs are currently mined and processed. This also includes a historical perspective on REE production, which allowed us to identify trends which could give insights in the future production of REEs. From a supply chain perspective, the most important mining and processing companies of REEs were also explored.

To answer the second research question, it is also crucial to assess the environmental impact of REEs. This is done by describing the situation and pollution around the mines and impact on the environment. Furthermore, the impact was also quantified. For example, through the examination of research which calculates the carbon emissions per kg of REE production, the wastewater per kg of REE production etc.

2.1. Rare-earth elements in wind turbines

This part was started with investigating research which explains which type of wind turbines contain REEs and why REEs are used in wind turbines. Similar to mining companies of REEs, manufacturing companies of the REE components in wind turbines were also investigated from a supply chain perspective.

Next, we analysed the share of the cost of rare-earth elements in the total LCOE. For this, we investigated how much REEs are used in a wind turbine, in kg per MW. This was then multiplied by the price, in order to compute the cost of REEs per MW. Given the net present energy, which was obtained for the LCOE, we know the discounted amount of MWh per MW. This way, the cost of REEs per MWh could be computed. This cost can also be compared to the LCOE, which is the total cost per MWh. Next, a sensitivity analysis is done based on the most important REEs in a wind turbine. In 2011, REE prices skyrocketed because of export restrictions in China. That's why we also estimated the cost of REEs per MWh based on the peak prices in 2011, to investigate whether disruptive shocks in supply or demand could meaningfully impact the use of REEs in wind turbines.

The next analysis is estimating the carbon footprint of the REEs used in a wind turbine. For this, we used the weight of CO_2 and equivalents which is generated per kg of REE production. Based on the amount of REEs used per MW, the grams CO_2 per kWh could be computed. Next, this is compared to the total carbon footprint of wind turbines, which is estimated for research question 1. Based on this, we could see if the use of REEs impacts the carbon footprint of wind turbines.

Furthermore, we also investigated potential substitute materials, recycling options, demand projections etc. in the existing literature.

2.2. Critical elements in solar energy

We considered thin-film PV panels for critical elements study, particularly CdTe and CIGS panels. Thin-film technologies use critical elements that are not REEs, however they are critical for future wide scale implementation for these technologies. Although thin-film technologies are only 5% of overall installed PV capacity, their share is expected to increase in the future due to efficiency increases and expected cost reductions in the future. We analysed supply chain risks of these elements based on 4 criteria (concentration risk, risk of demand increases etc.) and 11 indicators based on these criteria (recycling rate, future technology demand, substitutability etc.). Next, based on weighting of criteria and indicators, we came up with overall risk scores for these elements both in element level and technology level. For example, at technology level, indium is the riskiest for supply chain perspective because of low static reach reserves, low recycling rates and extraction as a by-product of zinc. At technology level, CdTe panels are less risky than CIGS panels.

Additionally, we investigated metal demand for PV panels and wind turbines in the future with the implementation of the Paris agreement on climate change based on a study conducted for Netherlands. The results are that it is not possible to implement the Paris agreement by 2050 if the critical metal and REE production as well as recycling rates stays the same. The production needs to be increased several folds to meet the renewable energy needs. For example, neodymium and indium production need to be increased more than 7 times and 12 times respectively by 2050 from the total production levels of 2017 to meet the requirements of solar and wind energy alone.

Other aspects

All currencies are converted to euro to enhance comparability. These are converted at the rates reported in table 2.

	US Dollar	Brittish Pound	Canadian Dollar	Chinese Yuan
1 EUR	1,11	0,91	1,52	8,33

Table 2: Exchange rates used

III. Findings

1. Life cycle cost

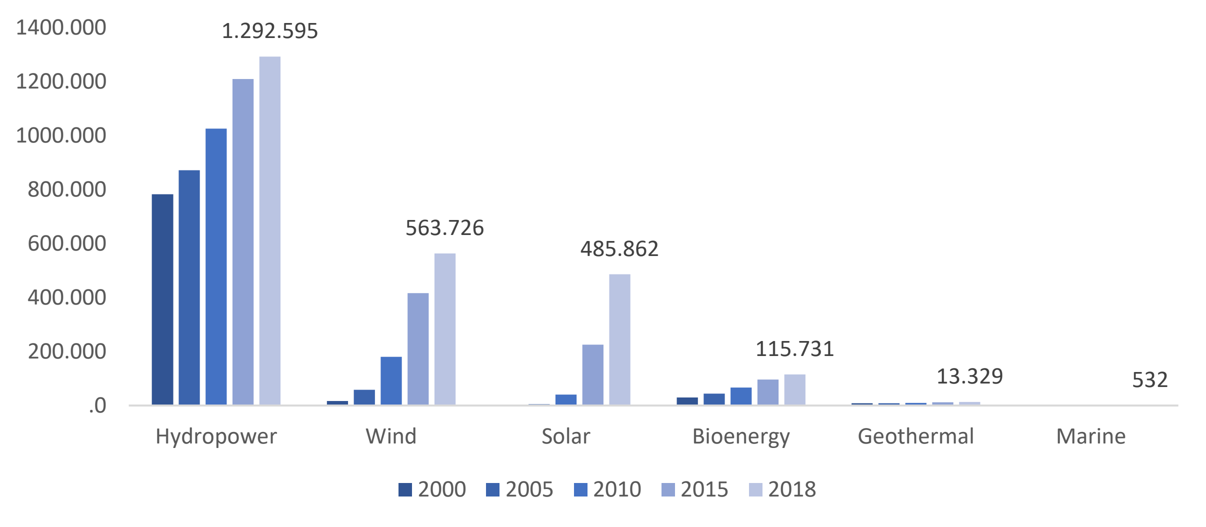

Figure 1: Evolution of installed capacity of renewable energy technologies in the world in MW (IRENA, 2017)

During the last couple of years, the installed capacity of mainly solar and wind energy has been increasing dramatically because of economies of scale, government subsidies and advancements in technology (Figure 1). The annualised growth rate of wind and solar energy between 2000 and 2018 has been 20% and 37% respectively (appendix 1). During that period, solar energy capacity exceeded bioenergy and geothermal energy and has been the fastest growing renewable energy technology (IRENA , 2018). Wind power has also been growing considerably and it was the second most widely used renewable energy source after hydropower. Hydropower itself has been the main source of renewable energy through the 18-year period, but the capacity growth has been slow compared to solar and wind with a cumulative aggregate growth rate (CAGR) of only 3%. In 2000, hydro installed capacity was already more than 2018 levels of wind and solar power (783, 004 MW). Given the popularity of solar and wind energy, this project focuses on these renewable energy technologies. Hydropower is also popular, but not relevant for Belgium.

LCOE-studies suggest that, in some cases, economic viability of renewables caught up and even exceeded traditional energy sources such as coal, nuclear or combined cycle gas plants (Lazard, 2018). In this part we will analyse the levelized cost of energy (LCOE) for wind energy and PV panels and, in the end, they will be compared with other energy sources. Furthermore, non-financial costs will be analysed by investigating recycling practices and carbon emissions.

1.1. On-shore wind energy

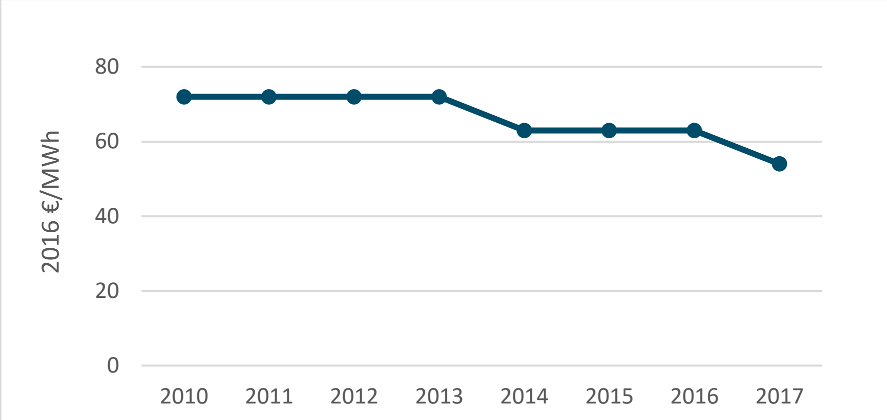

Figure 2: Weighted average global LCOE 2010-2017 (IRENA, 2019)

In recent years, the cost of wind energy has come down substantially, mainly driven improvements in technology and lower resource costs. Figure 2 shows the evolution of the weighted average global LCOE between 2010 and 2017, according to the International Renewable Energy Agency (IRENA). According to these data, the LCOE of onshore wind energy has dropped from 72 €/MWh in 2010 to 54 €/MWh in 2017.

There is a wide variety of data available about the LCOE for wind energy and substantial differences in estimates arise. While IRENA (2018) reports the average LCOE at 54 €/MWh, the range reported is between 36 €/MWh and 252 €/MWh. According to Lazard (2018), the global range for onshore wind energy is between 25 €/MWh and 50 €/MWh and between 38 €/MWh and 58 €/MWh for Northern Europe. It is clear that there is a lot of variation in cost across different sources, mainly caused by differences in assumptions and countries studied.

Furthermore, the studies available are often global studies or studies related to certain countries other than Belgium and many of these studies don't provide much insights into the components of the LCOE at each stage. We will look at the costs incurred at each stage and compute a range which is relevant for Belgium.

1.1.1. Installed cost

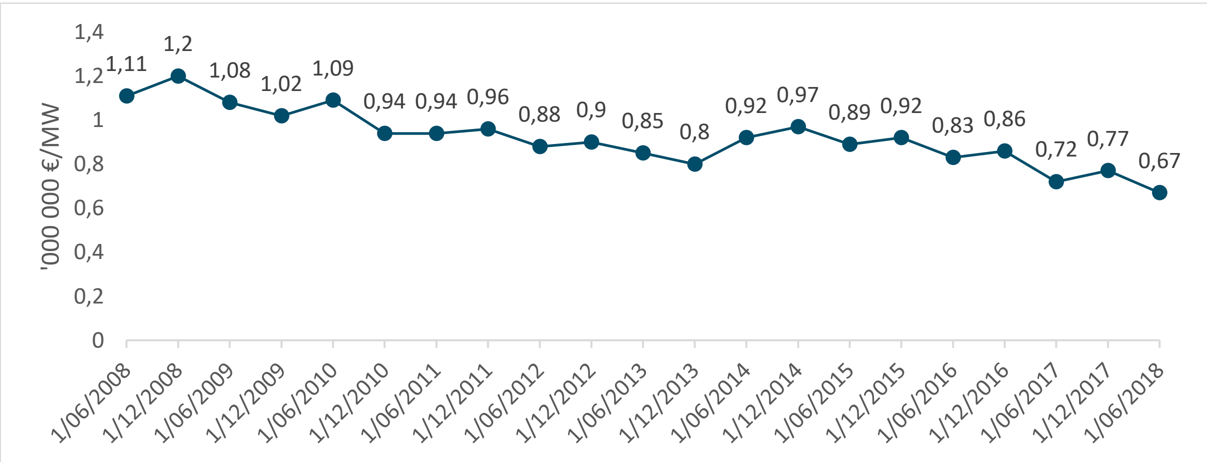

Figure 3: Wind turbine prices 2008-2018 (Bloomberg New Energy Finance)

Figure 3 shows that average cost of wind turbines almost halved between its peak in 2009 and the end of 2018. The downward trend could be explained by drops in construction costs and improved technology. Prior to the crisis in 2008, material costs (e.g. copper, steel) and labour costs have been rising. Since the financial crisis, drops in demand caused prices drops. Furthermore, due to improvements in technology, the nameplate capacity increased, which reduced the cost per MW. Other favourable factors are greater competition and economies of scale (IRENA, 2018).

The cost reported in Figure 3 includes the cost of towers and transport to the site. To put things into perspective, the average nameplate capacity of newly installed wind turbines in Belgium in 2017 was around 2,8 MW (WindEurope, 2018). Based on Bloomberg's global average cost, this means that the cost of a 2,8 MW wind turbine would be €1.876.000.

	% of total installed cost	€/MW ('000)	Implied cost 2,8 MW turbine ('000)
Turbine	74,53%	963,90	2.698,92
Rotor	21,15%	273,56	765,96
Nacelle	36,66%	474,07	1.327,40
Tower	16,72%	216,27	605,55
Balance of System	25,36%	327,60	917,28
Electrical infrastructure	11,41%	147,35	412,58
Assembly and installation	3,32%	42,92	120,17
Site Access and Staging	3,54%	45,78	128,18
Foundation	4,54%	58,65	164,23
Engineering Management	1,44%	18,60	52,07
Development	1,22%	15,74	44,06
Total installed cost	100,00%	1.292,93	3.620,21

Table 3: Installed cost breakdown (Stehly, Heimiller, & Scott, 2017)

While the turbine itself makes up the bulk of the total installed cost, there are other important costs which have to be incurred. Contracts typically include towers, installation and delivery. Furthermore, there are important costs, like planning and project costs, grid connection, the construction of access roads etc. Most sources report the initial installed costs, but don't go into detail about the cost components. Table 3 gives an overview of the total installed cost. These results are in line with the findings of IRENA (2017), who found that turbines typically make up 64-84% of total installed costs.

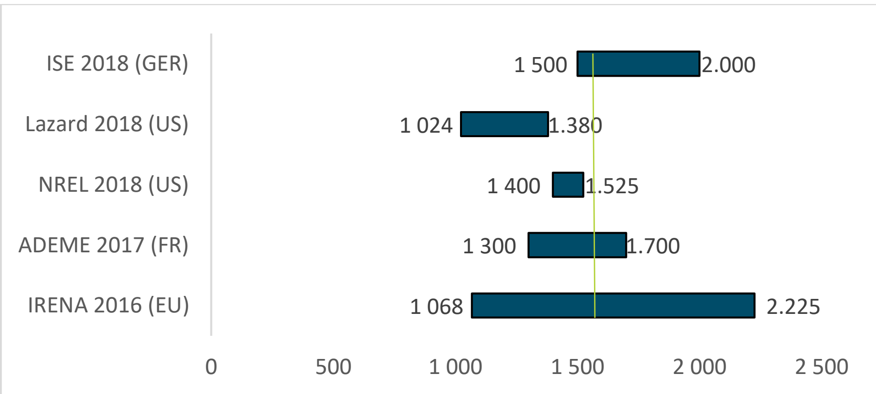

Figure 4: Overview estimates installed cost ranges in '000 €/MW (Kost et al., 2018; Lazard, 2018; NREL, 2018; ADEME, 2017; IRENA, 2018)

Figure 4 shows the ranges reported by various sources. It is clear that there is substantial variation in estimates across sources. There is also significant variation in installed costs across countries. For example, IRENA (2016) reports around double the installed cost of China for the UK. It also appears like installation costs are generally lower in the US compared to Europe. Based on the data in figure 4, we found a weighted average installed cost of 1.578.000 €/MW (appendix 2). Higher weights were given to European estimates, since our calculation concerns an estimate relevant for Belgium.

1.1.2. Operating and maintenance expenses

The installed cost is incurred at the beginning of the project. However, the lifetime of a wind turbine is typically 20-25 years. From the years following the installation until the end of life, there are still significant operating and maintenance expenses (O&M).

Figure 5 gives an overview of the O&M composition (Stehly et al., 2017). Most sources don't spit up O&M and report one single number. Moreover, while different sources were largely in line, we still found some. For example, maintenance generally makes up the largest proportion, but IRENA (2017) reports that maintenance makes up around 65% of total operating expenditures, instead of 55%. Also, while some sources (Stehly et al., 2017) report land lease costs separately, other sources include this into the cost of operations (IRENA, 2018). Furthermore, there are significant country-specific differences. For example, in France, IFER is included. This is a tax on certain network companies, and it makes up around 16% of operating expenses. This tax is mostly levied on companies active in energy and telecommunication (CRE, 2014).

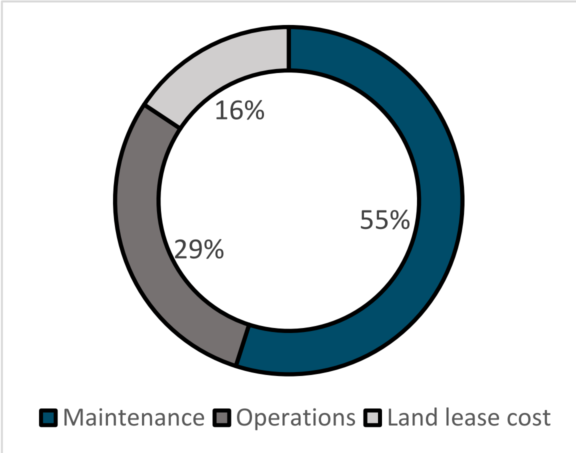

Figure 5: Overview O&M (Stehly et al., 2017)

Table 4 confirms that there's substantial variation across sources and countries. The lowest reported number is 14.000 €/MW, while the highest was 53.000 €/MW. A weighted average of 37.000 €/MW. Similar to the calculation of the installed cost, we gave higher weight to European sources.

Source	Year	Location	Low	High	Average	Weight
IEA (Hand, 2018)	2016	Germany	53	53	53	0,125
IEA (Hand, 2018)	2016	Europe	23	23	23	0,125
IRENA (2017)	2016	Europe	14	33	24	0,200
ADEME (2017)	2017	France	42	52	47	0,200
DOE (Wiser & Bolinger, 2018)	2017	United States	25	25	25	0,050
NREL (2018)	2018	United States	45	45	45	0,050
ISE (2018)	2018	Germany	39	46	43	0,200
Lazard (2018)	2018	United States	25	32	29	0,050
				Weighted average		37

Table 4: calculation weighted-average O&M expenses in '000 €/MW

1.1.3. End-of-life

On-shore wind turbines have an expected lifetime of 20-25 years. At the end of this period, there are 3 options: full repowering, lifetime extension and decommissioning (WindEurope, 2017). In case of full repowering, the wind turbines are dismantled and replaced with new wind turbines. Lifetime extension means that existing wind turbines are upgraded with new technology, but the overall external layout remains. If the wind turbines are not repowered or the lifetime is not extended, the turbine is decommissioned and sold for scrap. For our calculation, we will focus on decommissioning and assume a lifetime of 22 years, but also briefly discuss repowering.

Decommissioning

In some states in the United States, it is legally required for operators to maintain financial assurance in an amount equal to this net decommissioning cost for onshore wind farms, so it is certain that the expenses needed to restore the property at the end of its lifetime would be incurred by the operator (Cassadaga Draft Decommissioning Plan, 2017). These financial assurances could be met by contribution to a decommissioning fund or by posting a bond (Stripling, 2016). These costs of decommissioning estimates are often publicly available. Table 5 provides an overview of several of these estimates. A potential drawback of this method is that the operators have incentives to keep the estimation as low as possible, since this reduces the amount of financial assurances they have to maintain.

	MW	Total decommission cost (€)	Salvage value (€)	Net decommission cost (€)	Decommission cost per MW (€)
Stony Creek wind farm	1,50	n/a	n/a	40.027	26.684
Cassagada wind farm	2,63	149.934	146.288	3.647	1.389
McLean County Wind Energy	2,50	119.160	71.631	47.529	19.011
Dakota range wind project	4,20	213.318	176.993	36.324	8.649
Average	**2,71**	**160.804**	**131.637**	**31.882**	**13.933**

Table 5: Overview decommission costs per turbine for several projects (Decommissioning plan Stony Creek Wind Farm, 2009; GHD, 2017; Burns & McDonnel Engineering Company, 2018; Apex Clean Energy Management, 2017)

Table 5 shows that decommissioning costs are substantial and could amount to €213.318 per turbine. For decommissioning, the largest expense is the removal of the turbines. This is basically the reverse of the installation process. First the turbine blades, nacelle and the tower need to be removed and separated on-site. Also, the foundation needs to be removed, of which the depth typically extends to 7 to 15 meters. Other incurred costs are soil recovery and removal of power lines, access roads and transmission stations (Stripling, 2016).

While the total decommission cost is rather large, this is (partly) offset by the salvage value of the materials used in the turbines. These are mainly steel and copper. We found an average net decommissioning cost per MW of 13.933 €/MW. This average is assumed to be the decommissioning cost for the LCOE calculation. Furthermore, the costs generally only have to be incurred after between 20 and 25 years. Because of the time value of money, the present value of the cost is drastically lower. Given a discount rate of 4,56% and a lifetime of 22 years, the present value of the average decommissioning cost of 13.933 €/MW is only 5.228 €/MW. Compared to the average installed cost €1.578.000, this is almost negligible.

The estimates in Table 5 are based on the salvage value of the materials. There is the assumption of no resale value of the components. Table 5 includes the estimate for the Dakota range wind project. In the calculation, a separate scenario is developed where they calculate the net decommissioning cost, with the assumption that certain components could be resold (Apex Clean Energy Management, 2017). In that case, the total decommissioning cost remains at €213.318, but the salvage value increases from €176.993 to €236.861. As a result, there is a net gain of decommissioning of €23.554 per turbine. The authors also consider this scenario to be the most likely scenario, but for our calculation, we conservatively assume no resale value.

It is clear that the net decommissioning cost for on-shore wind energy is dubious and dependent on many variables. Even in case of conservative assumptions, such as no resale value, the average net present value of the net decommissioning cost is rather low. This might explain why the net decommissioning cost often is not mentioned in LCOE calculations.

Repowering

Apart from decommissioning, repowering is an alternative solution. According to certain sources, there could be substantial cost savings through repowering. A certain source even mentioned an installed cost reduction of 28,5% compared to a new-built site (Parsons Brinckerhoff, 2013). This cost saving would be realised through cost savings on planning, access roads which don't need replacement etc. However, alternative sources suggest that the costs of repowering are similar to newly built sites, because repowering requires a new planning application and new licensing. Furthermore, if the capacity of the turbines would be increased through the use of larger turbines, existing infrastructure might require replacement. For example, a new grid connection might be required because the capacity of the old infrastructure could be insufficient. Eventually, a potentially completely new layout might be due, and the potential costs and revenues would cancel each other out (Parsons Brinckerhoff, 2013). In conclusion, the cost savings due to repowering most likely vary substantially on project characteristics.

1.1.4. Capacity factor

WindEurope (2019) reported the average onshore capacity factor in Europe to 22% in 2018. However, this average is based on old and new turbines. More modern wind turbines have better technology and often higher hub heights. Because of this, they are more efficient. ADEME (2017) report capacity factors for France of 21-27% for standard wind turbines, but capacity factors of 27-31% for new generation wind turbines. Nonetheless, for the new generation wind turbines, the installed cost is higher. In Spain, a 21-year old wind farm was repowered, which resulted in an increase in capacity factor with 9%, from 22% to 24% (IEA Wind, 2017).

There is significant also variation across countries. For Belgium, the IEA Wind (2017) reported an onshore capacity factor of 22% in 2015, but 35% for Norway. It also appears like capacity factors in the United States are generally higher than Europe. In the United States, the average capacity factor in 2017 was 42% for projects built between 2014 and 2016 (Wiser & Bolinger, 2018). Lazard (2018) estimates a range of capacity factors between 38% and 55%.

JRC (2018) mapped capacity factors per country in Europe and found a capacity factor of 19% in Belgium for all available land. In practice, the capacity factor is probably higher, since operators will select locations with good wind conditions for the turbines. However, it is clear that the capacity factors in Belgium are relatively low, especially compared with other North-European countries. It appears like the capacity factor in Belgium is most similar to Germany, of which the same source (JRC,

2018) estimates the average capacity factor at 20%. Fraunhofer ISE (Kost et al., 2018) estimated capacity factors in Germany for several locations. Based on the average of Inland (21%) and Northern Germany (29%), we assume a capacity factor of 25% for Belgium.

1.1.5. Levelized cost of energy

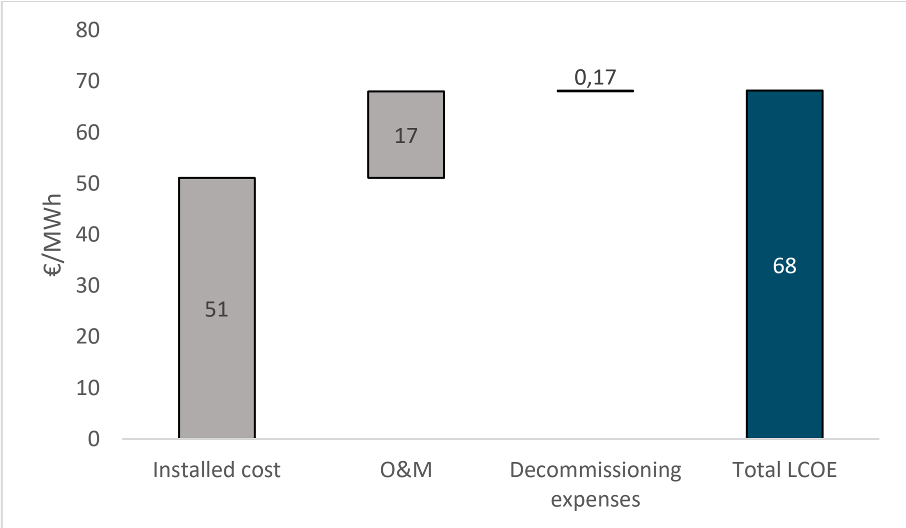

Figure 6: Contribution of each stage to the LCOE for onshore wind energy

Based on above-mentioned assumptions, we found a LCOE of €68/MWh. Clearly, the biggest part of the LCOE is the installed cost (74,9%). The O&M are also substantial (24,8%) and the decommissioning expenses appear to be almost negligible (0,2%). Appendix 3 also show sensitivity analyses for our main assumptions: the discount rate, capacity factor, installed cost and O&M.

1.1.6. Recycling

The decommissioning cost estimates mentioned earlier already provide insights into the value of scrap materials. The recycled materials are estimated at salvage value, whereas the non-recycled materials at disposal cost (e.g. landfill). For the decommissioning cost estimate of Cassadaga wind farm, it was assumed that 80% of the tower (steel) is recyclable, 100% of the hub (steel) and 80% of the nacelle, of which 5% is assumed to be copper. They don't assume any salvage value for the rotor and blades, since these composite materials consist mainly of fiberglass reinforced epoxy (FRP) and carbon fibres. They also assume a $1.000 disposal cost per wind turbine for disposal of petroleum, oil and lubricants. While there is less copper used in the wind turbine, the scrap value is almost 20 times higher than steel. For the Cassadaga wind farm, a scrap value of €211,5 per US ton is assumed and €3.762 per copper. As a result, the total value of scrap steel and copper was around the same level (GHD, 2017).

For the Dakota range wind project, similar assumptions were made. They make a distinction between low-, medium-, high-grade and reusable components. Low-grade materials, such as concrete rubble, wood, general waste etc., are assumed to be sent to landfill or incineration at a certain cost. They also consider FRP from the blades and rotors as waste. Medium-grade materials are such as cabinets, lighting, small motors etc. are sent to salvage centres and potentially resold, but if they can't be resold, they would also be sent to landfill. High-grade materials are similar to the salvageable materials in the Cassadaga wind farm estimate, but also include aluminium. They consider reusable components to be components such as electrical infrastructure and recently replaced turbine components (Apex Clean Energy Management, 2017).

According to Fraunhofer IWES, the recycling quota for wind turbine blades is over 80 or 90%, because there are well established return and recycling systems for components such as electrical parts, steel and concrete (Albers & Greiner, 2013). There are substantial environmental benefits of recycling these materials. If these materials are recycled, there is no more need to mine them and eventually less carbon is energy is required. Recycling steel requires 56% less energy compared to mining ore and it conserves natural resources, since they don't have to be mined again (West, sd). The recycling also reduces lifetime carbon emissions from 17,35 gCO_2/KWh to 9,78 gCO_2/KWh (Guezuraga, Zauner, & Pölz, 2012).

For certain types of wind turbines, there are also certain rare-earth elements which have to be considered (cfr. Infra).

Turbine blades

The biggest issue is that the turbine blades are generally not recycled. These blades are mostly made of FRP, which is low in cost and weight and high in strength and stiffness. All decommissioning estimates used to calculate the decommissioning cost, assumed no recycling of turbine blades. This assumption is in line with academic research (Beauson, Bech & Brøndsted; 2014). As the turbine blades are getting bigger and bigger, the FRP per turbine also increases drastically and when landfilled, harmful chemicals could be leached in the ground. The amount of FRP in wind turbines is typically 10 to 15 tonnes/MW and the use of FRP in blades was 150.000-186.000 tonnes in Europe (Jensen & Skelton, 2018).

In the US, blades are typically sent to landfill, as this is the cheapest solution (EWEA). In Europe, most turbine blades are incinerated, but in this case, up to 60% is left behind as ash (Jensen & Skelton, 2018). However, there are alternatives.

Beauson et al. (2014) identify several options to solve this issue. The first one is to refurbish the whole blade and then reuse it in another wind turbine. An advantage of reuse is that they are around half of the cost per MW compared to new blades. However, they found this to be limited to smaller blades, since there would be transportation issues for blades longer than 45m. The other option is to cut the blade in smaller pieces and use it in new applications, for example as a playground for children. When this is not possible, it is possible to cut it in even smaller pieces and use it for construction, for example as planks or plates. If there is no use for these, the blades could be either shredded or the glass fibre fabrics could be extracted. In case of shredding, the material could be used as a filling material (e.g. insulation) or in new composite material. The other option, extracting the glass fibre, hasn't really been implemented on a commercial scale, since the quality of the fibres reduces in the process and the value of the recycled material is relatively low. Shuaib & Mativenga (2016) found that recovering products from FRP composite waste requires only one tenth of the energy of virgin material, but also point to a significant reduction in quality.

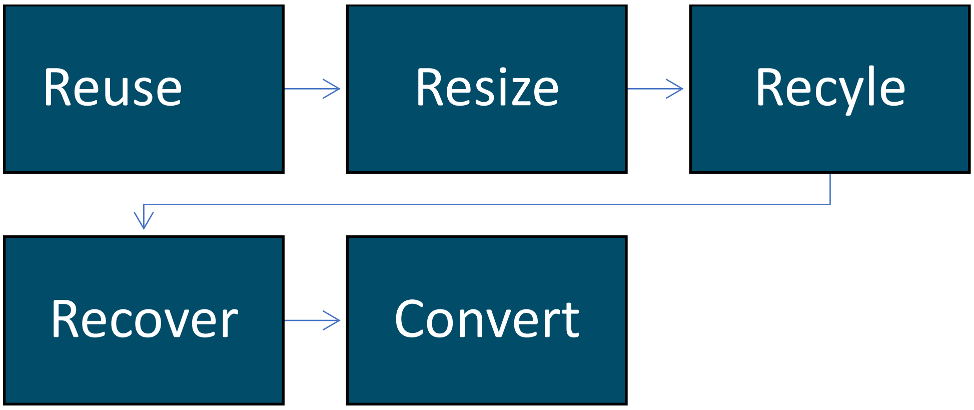

Figure 7: FRP end-of-life options (Jensen & Skelton, 2018)

Figure 7 shows the recommended end-of-life steps for turbine blades according to Jensen & Skelton (2018). Reuse is the preferred option, followed by resizing. This similar to Beauson et al. (2014). Recycling includes shredding and crushing and using this as, for example, a material in wood paint. For recovery, they consider pyrolysis and solvolysis, through which the fibres could be recovered. This way, the fibres could be, for example, used in concrete. The drawback is that this is complicated, often still in laboratory scale, and it merely substitutes a cheap material. For conversion, the material could be converted into valuable chemicals. Researchers succeeded to turn the FRP material into an oil with a caloric value similar to bio-oil. The drawback of this is that it is complicated, requires energy and is still in laboratory scale, but the technology seems promising.

Hoefer (2015) points out that significant increases in technology or government subsidies are required for the option of recycling these blades to be economically viable for the owners of the wind farm. Once this is the case, the decommissioning costs will go down and more blades will be recycled. WindEurope (2017) also mentions promising technologies, such as using chemicals which can recover fibre materials with similar strength as the original fibres. However, this technology is still at laboratory scale.

1.1.7. Carbon emissions

Stage	Share in energy requirement
Manufacture	84,4%
Transport	7,0%
Maintenance	4,3%
Operation	1,2%
Dismantling	3,1%

Table 6: Energy requirement per stage (Guezuraga, Zauner, & Pölz, 2012)

While wind turbines are often considered carbon neutral, this is not the case when the manufacturing and logistic processes are included. Table 6 provides an overview of share of each stage in the lifetime of a wind turbine in the total energy requirement (Guezuraga, Zauner and Pölz, 2011). The manufacturing stage represents, with 84,4%, the biggest energy requirement. In this stage, the construction of the tower makes up the biggest part, with 55%. Guezuraga et al. (2011) also found that the emissions are on average 9 gCO_2/kWh and the total energy requirement is paid back after 7 months. This is in line with Schlömer et al. (2014), who reported a range between 7 and 56 gCO_2/kWh, with a median of 11 gCO_2/kWh. However, the impact of carbon emissions is also highly dependent on the type of land on which the wind farms are located. For example, Thomson & Harrison (2015) found that the cost could be over 60 gCO_2/kWh when located in forested peat lands. The carbon emissions are still relatively low compared to other energy technologies. For example, the lifecycle emissions of coal range between 740 and 940 gCO_2/kWh (appendix 4). Other factors which impact the lifetime carbon emissions are lifetime, capacity factor, the wind turbine model and system costs.

1.1.8. Balance of system

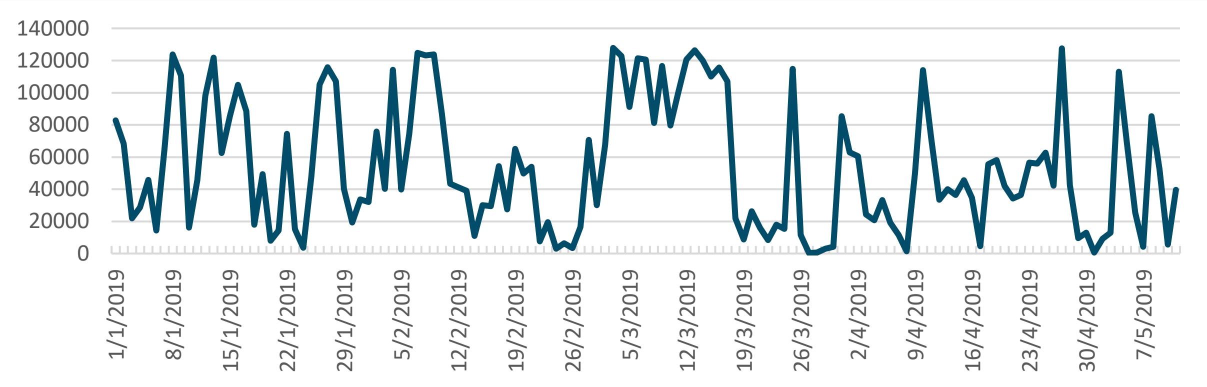

Figure 8: Elia's wind energy production per day 1/1/2019 - 12/05/2019 (Elia, 2019)

Another important issue for wind energy is large swings in energy production, which is illustrated by figure 8. For example, from 22/02/2019 until 22/02/2019, there wasn't a single day when more than 20.000 MWh was produced through wind energy, but during March, there were days when more than 120.000 MWh was produced in a single day. These swings in energy production make wind energy a less reliable source of energy and, even if wind energy would be the cheapest energy source, it would almost be impossible to rely entirely on wind energy. As a result, the energy generated could be considered less valuable than energy generated by more predictable energy sources.

The LCOE relates only to the cost accruing to the owner of the generation plant. Consequently, it measures the cost of generating the power and sending it to the grid. In the installed cost, the grid connection costs are considered, but the energy still has to be transferred to the end user. Furthermore, because of the variability in energy supply of wind turbines, there are substantial balancing costs. This means that the LCOE does not consider the impact it has on other generators and the system as a whole is excluded. The problem with estimating these system costs is that there is very little agreement about the amount, and they are dependent on several variables. It is dependent on network capacity, interconnection, generation mix and the availability of variability management mechanisms (Thomson & Harrison, 2015).

Cost component	€/MWh
Balancing	2,2 – 7,2
Backup	0,2 - 0,6
Transmission	5,5 – 11
Total system costs	7,7 – 19,8

Table 7: System costs (Thomson & Harrison, 2015)

These system costs are relevant for all energy technologies, but the main difference of wind energy compared to other technologies is higher balancing costs. Because of the variability in supply as shown in figure 8, flexible reserves are needed to handle unpredictable peaks in demand or wind generation. These reserves are power stations which are running at part load and standby generators which can be turned on rapidly. The need to have standby generation and plants which operate less efficiently raises a certain cost which wouldn't be there if wind was more predictable (Thomson & Harrison, 2015). These costs are difficult to estimate because they become higher when the share of wind in total electricity generation increases. Furthermore, the carbon emissions of these power plants used for balancing are also substantial. Table 7 provides an estimate of the system costs.

1.2. Offshore wind energy

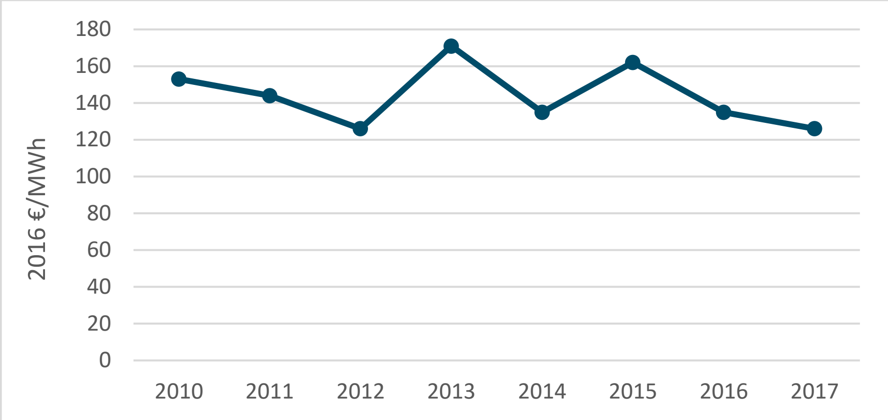

Figure 9: Weighted average global LCOE 2010-2017 (IRENA, 2019)

Figure 9 shows that the global weighted average cost of offshore wind energy has come down with 18% since 2010, from 153 €/MWh in 2010 to 126 €/MWh in 2017. However, the 140 €/MWh in 2017 is still substantially higher than the 54 €/MWh which IRENA reported for 2017 for onshore wind energy.

While this global average is of offshore wind energy is more than double the global average for onshore wind energy, the installed capacity of offshore wind energy has considerably increased in Belgium. In 2018, the United Kingdom and Belgium were the only countries in the Europe with more new offshore installations than onshore and 35% of all wind turbines are installed offshore in Belgium (WindEurope, 2019). Given the generally higher nameplate capacity and capacity factor of offshore wind turbines, we could assume that considerably more than 35% of wind energy in Belgium is generated through offshore wind turbines. Also, because of Belgium's relatively high share in offshore wind turbines, more data for Belgium specifically is available.

It seems counter-intuitive that the cost of offshore wind turbines is globally substantially higher than onshore wind turbines, but Belgium chooses to install significantly more new offshore turbines compared to onshore. This indicates that the cost difference might not as big for Belgium. Other reasons are also possible, such as less availability of attractive onshore sites as wind is being more employed onshore, resistance from the local population (Hevia-Koch & Jacobsen, 2019) or the 95 €/MWh Renewable Energy Certificates granted by the Belgian government for 20 years on top of the wholesale price (Noonan et al., 2018).

There are 2 main kinds of off-shore wind turbines: fixed-bottom wind turbines and floating turbines. As the name suggest, fixed-bottom wind turbines are fixed to the ground, most commonly through monopiles. However, the monopile design reaches its engineering limits at a water depth of around 30m. More expensive jacket foundations are economically viable until water depths of around 50m (Myhr, Bjerkseter, Ågotnes & Nygaard, 2014). Recently, floating wind turbines have been developed. These allow wind turbines to be placed at deeper water depths and allow access to sites with more favorable wind conditions. In 2019, most off-shore wind turbines are fixed-bottom, as it is currently the cheapest kind of wind turbine (ADEME, 2017). For this reason, we focus on fixed-bottom wind turbines. This analysis will provide more insights into the lifetime cost of offshore wind energy.

1.2.1. Installed cost

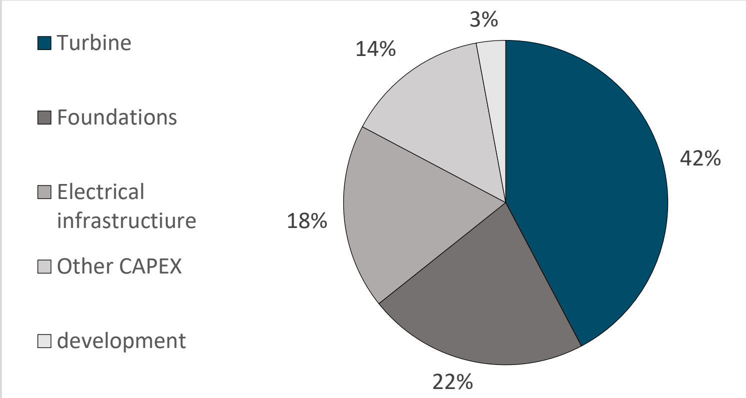

Figure 10: Installed cost breakdown (Noonan et al., 2018)

Figure 10 provides an overview of the installed cost components according to the International Energy Agency (IEA). While wind turbines still make up the largest cost component, the share in total installed cost is substantially smaller than for onshore wind turbines. According to IRENA (2018), wind turbines typically make up 30-50% of total installed cost, as opposed to 64-84% for onshore wind turbines. Another remarkable difference is that the share of foundations in total cost is now 22% (figure 10). For onshore wind farms, this was only 4,45%. Appendix 5 provides a more detailed overview of the costs typically incurred for an offshore wind turbine park.

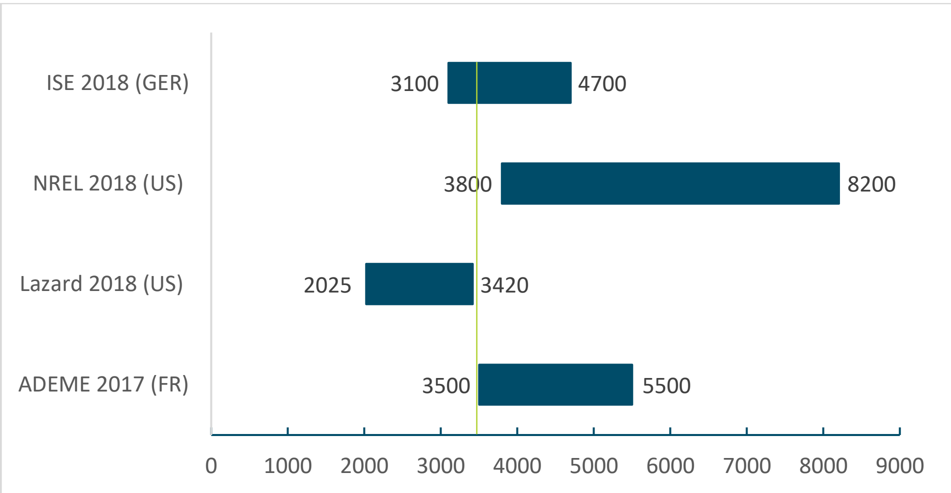

Figure 11: Comparison installed costs in '000 €/MW (Kost et al., 2018; NREL, 2018; Lazard, 2018; ADEME, 2017)

Figure 11 shows estimates of installed costs of several sources. The green line (3.458.000 €/MW) represents the IEA's estimate for Belgium (Noonan et al., 2018). We use this estimate as our input variable for the LCOE calculation. This estimate also seems to be in line with other sources. Furthermore, some estimates include some form of decommissioning costs into the initial capital expenditures, for example as a financial cost for a bank guarantee. Noonan et al. (2018) doesn't include this, so by assuming this estimate for the LCOE, so double counting is avoided when decommissioning costs are included.

Offshore wind farms are drastically more expensive for several reasons. First, planning and construction are more complex. This is illustrated by the increased cost of foundations. The costs of protecting the equipment are also higher compared to onshore wind projects due to the rougher marine environment. In addition, grid connection costs are higher, as offshore wind turbines are located further from the port of installation and the need to deploy undersea cables (IRENA, 2018).

1.2.2. Operating and maintenance expenses

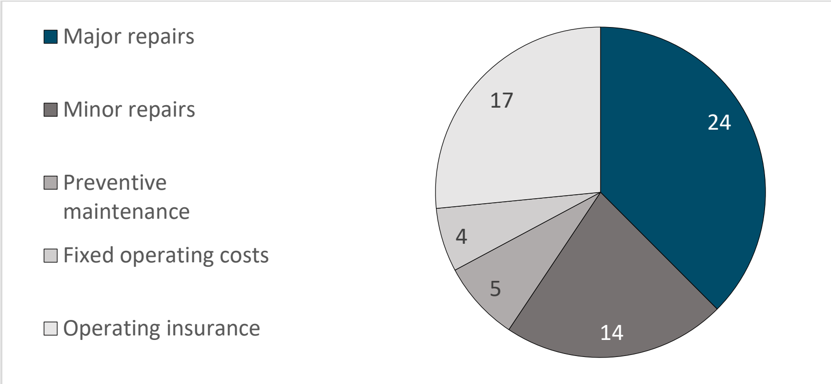

Figure 12: O&M costs breakdown for Belgium in '000 €/MW (Noonan et al., 2018)

Figure 12 provides an overview of O&M costs for Belgium. This is estimated to be €66.000 per year (Noonan et al., 2018). Figure 12 shows that the largest share of costs, 21.000 €/MW, are operating and the other 43.000 €/MW are maintenance costs. As a result, around 67% of costs are maintenance costs, similar to the 65% maintenance cost reported by IRENA (2017) for onshore wind turbines (cfr. supra). However, the €66.000/MW is considerably higher than the 37.000 €/MW, which we assumed for onshore wind turbines.

There are several reasons why O&M costs are generally higher for offshore wind farms compared to onshore. The main causes are higher access costs to the site and higher costs of performing maintenance of towers and cabling. It is also harder to operate in a marine environment compared to dry land (IRENA, 2018). Moreover, large differences in O&M across offshore projects could arise due to the local characteristics of the site. The 2 largest cost drivers are distance to the maintenance facilities and meteorological ocean climate at the site (Stehly et al., 2017). This is also illustrated by the increase in LCOE in 2013 (figure 9), as wind turbines were located farther from the shore, causing the distance to maintenance facilities to increase (IRENA, 2018).

1.2.3. End-of-life

Decommissioning offshore wind farms is drastically more expensive than decommissioning onshore wind farms. As mentioned for onshore wind farms, the decommissioning process could be seen as the reverse of the installation process. Given that the installation process of onshore wind farms is more expensive, this is also the case for the decommissioning process. Smith, Garret & Gibberd (2015) estimate decommissioning costs to be 60-70% of installation costs.

Arup (2018) estimates total decommissioning costs for 37 offshore wind farms to be within a range of €1,41 billion and £4,00 billion. Given that offshore wind turbines are often owned by shell companies with limited liabilities, it makes sense that governments are worried about the decommissioning of these offshore wind turbines. Arup (2018) estimates that the UK Crown Estate and The Scottish Government are potentially liable to a balance between €1,13bn to €3,23bn in case of default. However, given that decommissioning costs are incurred at the end of life, the cost today is relatively small and Arup (2018) estimates the impact on the LCOE to be less than 1%.

Smith et al. (2015) estimate decommissioning costs around 200.000 €/MW for some older wind farms, with small turbines located in shallow waters close to the shore. In these locations, the wave heights are relatively low, thus requiring relatively small crane vessels. For other projects, decommissioning costs range between 300.000 €/MW and 500.000 €/MW. These costs are higher because of higher winds, deeper water, higher waves and further distances to the shores. However, these estimates don't consider the recycling revenue, as "the quoted decommissioning costs may not include any offset anticipated from the revenue from materials recycling or r-sale of components" (Smith et al., 2015). As a result, in practice, the net decommissioning cost will be lower than these estimates. Topham & Mcmillan (2016) found similar results, with average decommissioning costs over €220.000.

	Low	Base case	High
Total decommissioning cost	315.700	436.040	555.720
Salvage value	24.200	24.200	24.200
Net decommissioning cost	291.500	412.060	531.520

Table 8: Overview decommissioning costs per MW of nameplate capacity in € (Smith, Drunsic, Reynolds, & Whitmore, 2016)

Table 8 shows an overview of the decommissioning estimates of DNV GL for the Canadian government of Ontario (Smith et al., 2016). These results follow detailed calculations, are in line with the estimates of Smith et al. (2015) and include salvage value. We use the base case estimate for the calculation of the LCOE.

Note that the estimated salvage value in table 8 is substantially lower compared to the estimated salvage values estimated for onshore wind turbines. The reason for this is that lower scrap values were used because of the assumption that the point of quayside delivery to the industrial recycling companies. This means that it is assumed that the recycling company bears cost of breaking the components down to separate the individual materials etc. As a result, the decommissioning cost is reduced, as well as the salvage value, thus both elements offset each other. This assumption wasn't made for onshore wind turbines, resulting in higher salvage values.

The decommissioning costs for offshore wind turbines are substantial. Table 8 only provides an overview of the cost per MW, however decommissioning costs after salvage value for the entire wind farm are as high €123.618.000. Climate Change Capital (2010) found the middle life accrual method to fund the decommissioning of wind turbines as the preferred method. This means that at the middle of the lifetime, the owner of the wind farm starts saving money for decommissioning. However, this study was withdrawn because the assumed decommissioning costs of 44.000 €/MW were outdated, and other research showed higher drastically higher decommissioning costs. Because of the higher decommissioning costs than anticipated when recommending the middle life accrual method, it could be a sign that it is optimal to start saving money for decommissioning earlier.

1.2.4. Capacity factor

As mentioned earlier, almost all costs for offshore wind turbines are higher compared to onshore wind turbines. However, these higher costs are (partly) offset by higher capacity factors due to the availability of better wind resources, less turbulence and steadier winds (IRENA, 2018). In 2017, the global weighted average capacity factor was 39% for offshore wind (IRENA, 2019), and 42% for newly commissioned plants (IRENA, 2018). For Belgium, the capacity factor is estimated at 42,94% for 2017 (Noonan et al., 2018). This is also the capacity factor we assume for the LCOE calculation.

1.2.5. Levelized cost of energy

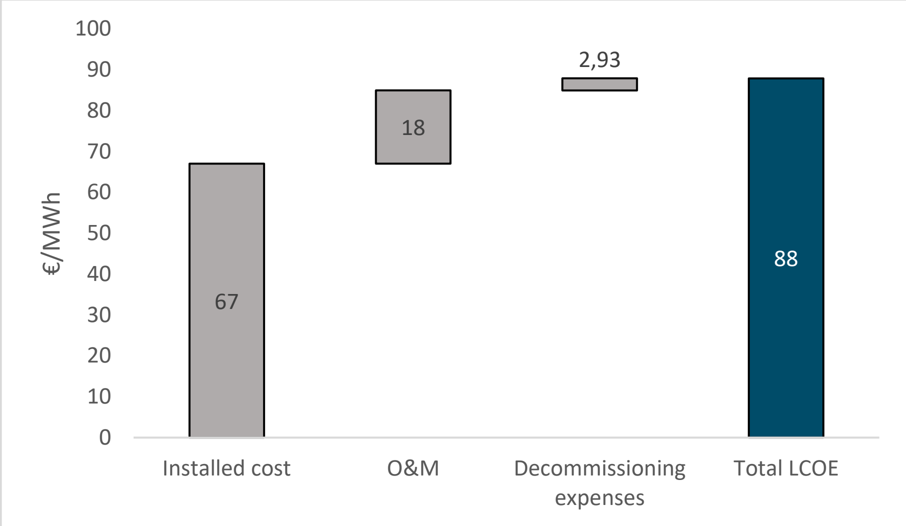

Figure 13: Contribution of each stage to the LCOE for offshore wind energy

As figure 13 shows, we found a LCOE of 88 €/MWh. As for onshore wind turbines, installed cost still makes up most of LCOE and O&M is still sizable. While decommissioning was negligible for onshore, it makes up 3,34% of total LCOE for offshore. This is considerable and it is important to take this into account for the calculation of the cost of offshore wind turbines.

Our finding of 88 €/MWh is substantially lower than the LCOE calculated by IRENA (2017) of 126 €/MWh in 2017 (figure 9). One of the main reasons for this is the lower discount rate used. If we use the same discount rate of 7,5%, we find 105 €/MWh. We also assumed lower installed cost and O&M. Another reason is improvements in technology and relatively close access of Belgian wind farms to shore. Appendix 6 provides sensitivity analyses for discount rate, capacity factor, installed cost and operating and maintenance expenses.

In our calculation, offshore wind still seems to be more expensive compared to onshore wind, but the gap seems to be relatively small compared to findings of other sources. For onshore wind energy, we found a LCOE of 68 €/MWh, which is 20 €/MWh more expensive than offshore wind energy. According to IRENA (2017), the difference is €72. The small difference we found might partly be due to the characteristics of Belgium. For onshore wind energy, the capacity factors are substantially lower compared to other locations (e.g. Scandinavia, US), but for offshore wind energy, Belgium's capacity factors are much more competitive. Another reason for the relatively small gap in costs is our discount rate assumption. We assumed that the discount rate is the same for each technology, but some sources use different discount rates. Offshore wind farms are often considered to be riskier compared to onshore, and as a result, the discount rate used is sometimes higher. For example, in Germany, ISE (Kost et al., 2018), use a real WACC of 2,5% for onshore wind and 4,8% for offshore wind, which means that the real WACC used for offshore wind is 2,3% higher. An increase in discount rate from 4,56% to 6,86% to correct for the higher risk of offshore wind, results in a LCOE of 101 €/MWh. In this case, the gap between onshore and offshore wind increases from 20 €/MWh to 33 €/MWh.

Technology improvements could further close this gap and make offshore wind turbines more financially viable. For example, regarding floating wind turbines, there's still a lot of room for technology improvements. There will most likely be substantial learning effects from implementing this technology on a larger scale and as a result the LCOE will go down. For example, for floating wind turbines, Myhr et al. (2014) already found that the LCOE could be as low as 82 €/MWh for conceptual designs at an optimal location. Another factor which could close the gap is increasing acceptance costs for onshore wind turbines. As the most convenient locations to place wind turbines onshore will be taken and wind turbines will need to be placed in places with more resistance from the local population, the acceptance costs could increase. Hevia-Koch & Jacobsen (2019) found that, for Denmark, the clear cost-advantage of onshore wind turbines disappears in certain scenarios of high wind expansion, which causes acceptance costs to increase. In a densely populated country like Belgium, these acceptance costs are most likely high as well.

1.2.6. Recycling

The recycling practices for offshore wind turbines are the same as onshore wind turbines, with some differences. The main difference is that there is significantly more steel used for offshore wind turbines because of the foundations. Often, the foundations are cut 1-5m below the mudline underneath the seabed and the steel part above it can be recycled. Another difference is that in some cases it can be economically viable to dig up the cables connecting the wind park to the shore from underneath the seabed and recycle the materials used in them (e.g. copper). Since the distance to shore is often several tens of kilometers, a lot of materials can be recovered from these cables. However, the net revenue from digging up these cables will be rather small, because the revenues are canceled out by the costs (Smith et al., 2015).

1.2.7. Carbon emissions

Lifetime carbon emissions per MWh produced are typically higher for an offshore wind farm compared onshore wind farm. While research reports an average of 9 gCO_2/kWh for onshore, the carbon emissions of offshore range between 7-23 gCO_2/kWh (Thomson & Harrison, 2015). This is still substantially lower than non-renewable energy (appendix 4).

Stage	onshore	offshore
Manufacture	84%	± 70,0%
Transport	7%	
Maintenance	4%	± 20,0%
Operation	1%	
Dismantling	3%	± 6,0%

Table 9: Comparison of carbon emissions for on- and offshore wind energy (Guezuraga et al., 2012; Thomson & Harrison, 2015)

As Table 9 shows, most carbon emissions emerge during the manufacturing phase and the installation phase. This is typically 70% of total carbon emissions, of which the vast majority arise from extraction of materials and the manufacturing of the components. While this is high, it’s still substantially lower than onshore wind turbines, where 84,4% of carbon emissions emerge during manufacturing.

The carbon emissions during other stages are significantly higher. This is because there are substantial carbon emissions during transport because of the use of vessels, but we found no explicate estimates of this in the share of total carbon emissions. Another reason is higher carbon emissions during O&M, which are estimated to be 20% of total carbon emissions (Thomson & Harrison, 2015). That is drastically higher than the share in onshore wind turbines of around 6%. The reason for this is that the turbines are more difficult to access. For example, helicopters are often used to access them. The decommissioning stage is estimated to make up 6% of the carbon emissions, which is also higher than the estimated 3,1% of onshore wind turbines (Thomson & Harrison, 2015).

1.3. Solar energy

Photovoltaic panels (PV panels) convert sunlight into electricity using semiconductor materials. There are several different types of solar panel technologies and there are 2 main categorisations.

Firstly, solar panels can be categorised as single-junction and multi-junction. Single junction technologies use just one layer of semi-conductive materials to absorb sunlight and convert it to electricity through the photovoltaic effect. Multi-junction technologies use different layers of semi-conductive materials to take advantage of different wavelengths of sun's energy.

Secondly, there is the categorisation of first, second and third generation technologies. First generation of solar panels include silicon solar cells. They are either made of a single silicon crystal (mono-crystalline) or cut from a silicon block that is made of many silicon crystals (multi-crystalline). Mono-crystalline panels are generally slightly more efficient because of the use of a single, pure silicon, but they are also slightly more expensive because of a more complex manufacturing process. The lower efficiency of multi-crystalline panels is offset by the lower price and therefore, they are competitive with mono-crystalline panels. Second generation of solar technologies are thin film technologies, which are less expensive to produce, require a lower amount of materials, but are generally less efficient than crystalline silicon technologies. They are made by replacing the silicon with one or more layers of films of photovoltaics. There are several types of thin-film solar panels available: amorphous Silicon solar cells (a-Si), cadmium telluride/cadmium sulphide solar cells (CdTe/CdS) and copper indium gallium selenide (CIGS) solar cells. There is also gallium arsenide (GaAs) solar cells that are very efficient, but extremely expensive and mainly used in space technologies such as satellites. Third generation solar cells are mainly in research and development stage and yet to be proven commercially. They generally use thin film technologies using different semi-conductive materials (Bagher, Mahmoud, & Mirhabibi, 2015).

About 95 percent of photovoltaic cells installed worldwide are silicon based crystalline, that is mono-crystalline and multi-crystalline cells. Despite them being older technologies, they are a popular choice for manufacturers and users because of the efficiency and availability of silicon compared to other types of semi-conductive materials (Fraunhofer ISE, 2017).

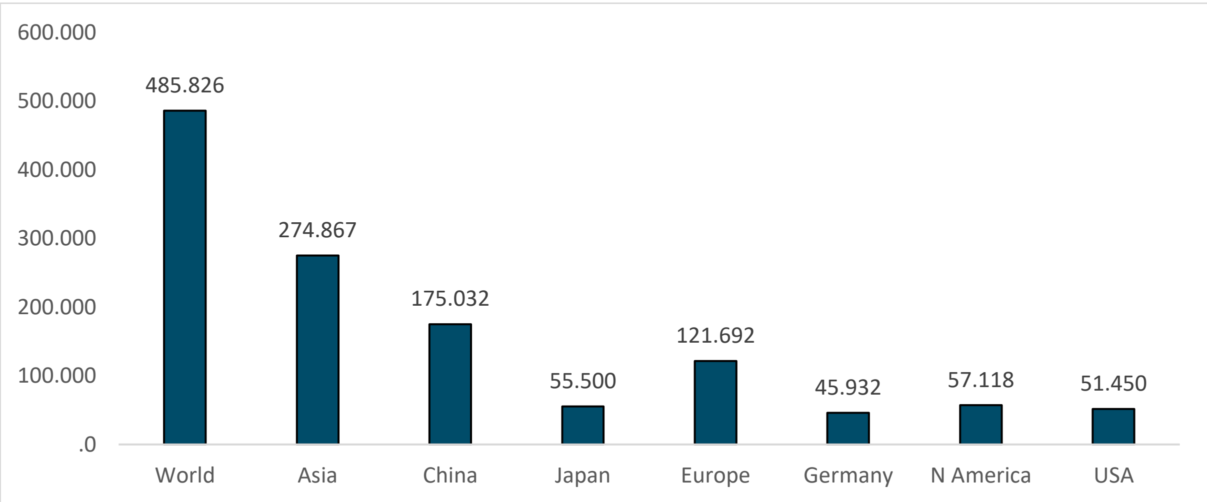

Figure 14: Installed cumulative PV capacity by continents and biggest regions for 2018 in MW (IRENA , 2018)

As of 2018, Asia makes up the majority installed capacity, with 56% of total installed capacity. In fact, China and Japan alone make up 48% of the world share. Europe has 25% share of the world capacity, of which Germany has biggest share with 9,5% of the world share. The next big region is North America with 11,8% of world capacity, of which 10,6% in the United States. Asia, Europe and North America together makes up 93,4% of world installed solar capacity (Figure 14). China's expansion is the fastest in the last several years. From 2010 to 2018 China's installed PV capacity share in the world rose from 2,5% to 36% (from 1,025 MW to 130,816 MW) and this corresponds to CAGR of 190,1 % (Appendix 1). Moreover, appendix 7 provides an overview of the installed capacity by region from 2010 to 2018.

A study conducted by Fraunhofer Institute for Solar Energy systems (ISE) covers energy cost of renewable energy and conventional systems for Germany. We consider that it is a good proxy for Belgium in terms of installed cost, operational costs and energy irradiance (Fraunhofer ISE, 2018). Based on this study, we conduct variance analysis for different irradiance levels and cost of capital to find out how changing variables affect LCOE.

The study covers small rooftop solar systems (5-15 KW), Large rooftops (100 kW-1000 MW) and utility scale solar (>2 MW), as well as onshore wind (2-4 MW), offshore wind (3-6 MW), biogas (>500 kW) and conventional systems, such as coal, combined gas cycle, gas turbines and CSP Fraunhofer study covers small rooftop solar systems (5-15 kW), Large rooftops (100-1000 KW), utility scale solar (>2 MW), onshore wind (2-4 MW), offshore wind (3-6 MW) and biogas (>500 KW) as well as conventional systems of coal, combined gas cycle, gas turbines and CSP. We will focus on solar power in this section.

1.3.1. Installed cost

Depending on the energy source, Fraunhofer ISE (2018) calculates low and high levels of investments per MW of installed capacity. You can see from Table 10, only gas turbine's initial investments are lower than utility scale solar power systems. While initial investment per MW might be interesting, it cannot be considered a true comparison as it does not cover all costs and electricity generation of the system. LCOE is a much better tool to compare economic viability of different systems. For our LCOE, average capital costs of 700.000 €/MW are assumed, as this is the average of the low and high estimate for utility scale PV.

Alternative	Low	High
PV small rooftop (5-15 kW)	1.200	1.400
PV large rooftop (100-1000 kW)	800	1.000
PV utility scale (>2MW)	600	800
Wind onshore (2-4 MW)	1.500	2.000
Wind offshore (3-6 MW)	3.100	4.700
Biogas (>500kW)	2.000	4.000
Conventional	Low	High
Brown coal	1.600	2.200
Hard coal	1.300	2.000
Combined cycle gas turbine	800	1.100
Gas turbine	400	600

Table 10: Installed cost for different energy technologies in '000 €/MW

The installed cost has also come down a lot historically. From 1976 to 2017, the price of crystalline silicon modules fell from 71 €/W to 33 €/W. This corresponds to a learning rate[2] of 28,5%. It is expected

[2] The learning rate is calculated as the cost reduction per doubling of deployed capacity.

that, in the next 17 years, capital expenditure of PV plants will halve (Bloomberg New Energy Finance, 2018).

1.3.2. Operating and maintenance expenses

O&M never received much attention for solar PV, since they have historically been very small compared to the installed costs. However, since the installed costs has come down substantially over the years, the share of O&M in the total LCOE has increased. Fraunhofer ISE (2018) now estimates these costs at 2,5% of the installed cost. However, these costs have to be incurred yearly, whereas the installed cost only once.

According to IRENA (2018), maintenance costs generally make up 45% of total O&M. Other costs are land lease costs, site security, administration etc. Mainly land lease costs are found to be very dependent on the location. For example, in a desert, they would be rather low, but in densely populated countries like Belgium, the will be higher. We used the 2,5% estimate of Fraunhofer ISE (2018) as our proxy for Belgium.

1.3.3. Irradiance levels

According to a websites that shows GHI (global horizontal irradiance) levels all over the globe, the average GHI in Belgium, is between 1.050 and 1.100 kWh/m2, which translates to a range between 1.011 and 1.065 MWh per MW (globalsolaratlas.info, 2019). Fraunhofer ISE (2018) mentions 950 GHI for Northern Germany and 1.300 GHI for Southern Germany. Our base LCOE calculation involves average GHI of 1.125 kWh. This is the average of Northern and Southern Germany according to Fraunhofer ISE (2018). This corresponds to 1.105 kWh of electricity generation (PVOUT), or a capacity factor of 12,6%.

1.3.4. Levelized cost of energy

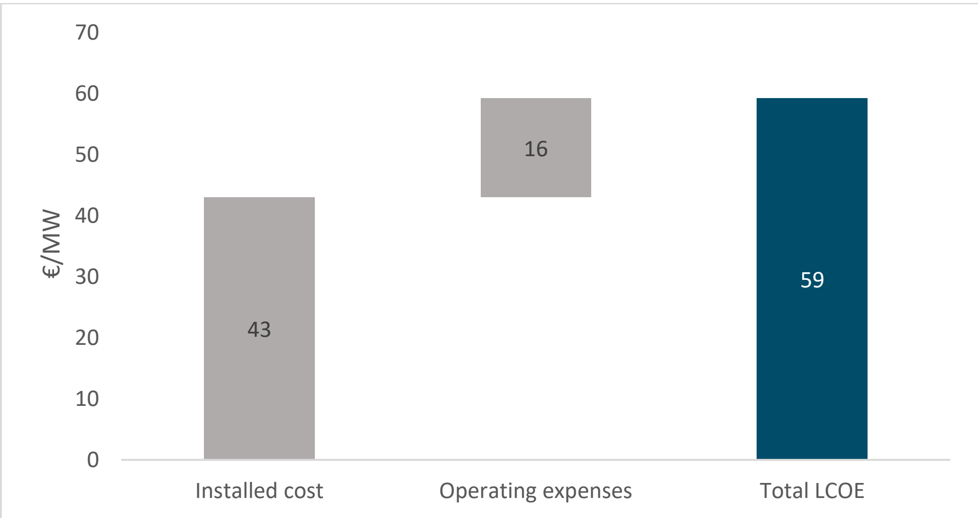

Figure 15: Contribution of each stage to the LCOE for solar energy

Our base LCOE calculation is 59 €/MWh. A sensitivity analysis of LCOE is also included, similar to what we did for wind turbines (appendix 9). This is done for discount rate, irradiance levels, installed cost and O&M. Both installed cost and O&M are slightly lower than wind energy, but the results are very comparable. For both technologies, most of the costs are the installed costs and O&M are much smaller. Utility scale solar appears to be the now the most competitive energy source.

Irradiance levels greatly affect the LCOE of solar PV systems. For example, changing electricity generation capacity[3] from 800 to 1.600 MWh per MW changes the LCOE from 79 to 39 €/MWh respectively.

Additionally, the lifetime of PV systems is expected to increase. Already many manufacturers guarantee their module performance over 25 years. It is expected that a 5-year increase in lifetime from 25 to 30 years of PV solar systems will result in decrease of LCOE with 5% to 56 €/MWh. Operating costs have a rather small influence on LCOE because they constitute a smaller share of total costs and unlike initial investments, they are incurred over the project lifetime and expenses in the future have lower impact due to discounting. An increase in lifetime has a strong sensitivity to LCOE because plants which are already fully amortized continue to produce electricity at very low operating costs.

European Waste Electrical and Electronic Equipment Directive (WEEE), under the notion of extended producer responsibility, dictates that producers, regardless of the origin of production, are physically and financially responsible for disposal of the end-of-life PV panels. Therefore, these costs are not considered in LCOE calculations. However, parts or the full amount of these costs might be borne by customers indirectly as producers might increase selling prices to accommodate end-of-life costs (European Parliment, 2012).

CAPEX	Low	High	Low	High
Irradiance	Low (950 GHI)		High (1300 GHI)	
Small rooftop (5-15 kW)	98,9	115,4	72,3	84,3
Large rooftop (100-1000 kW)	67,7	84,4	49,5	61,8
Utility scale (>2MW)	50,8	67,7	37,1	49,5

Table 11: LCOE (€/MWh) of solar technologies in Germany with different CAPEX and irradiance levels (Fraunhofer ISE, 2018)

Table 11 shows the findings of Fraunhofer ISE (2018), which is the study on which we based our assumptions regarding installed cost, O&M and irradiance level. As a result, it is only logical that our result is in line with this study. It is shown that even with low levels of irradiance (950 GHI) and high level of initial investments, utility scale solar panels achieve a LCOE of 67,7 €/MWh. The total range is between 37,1 and 67,7 €/MWh

Fraunhofer ISE (2018) found utility scale solar to be the most competitive energy source among wind, biogas and other conventional energy sources. Onshore wind and utility scale solar are cheaper than all the conventional energy sources (appendix 10) and their LCOE is expected to fall even further.

Our estimate is also at the low end of the range of IRENA (2017). For PV panels, in 2010 the minimum LCOE is 54 €/MWh and maximum is 360 €/MWh, with an average of 324 €/MWh. In 7 years, prices went down considerably, and the average price is 90 €/MWh, with a minimum of €45 and maximum of 315 (appendix 11). Differences between minimum and maximum values mainly stem from differences in irradiance levels, cost of raw materials (such as silicon etc.) and cost of financing (IRENA (2017).

[3] This is a direct function of GHI. Our base case GHI is 1.125, resulting annual electricity generation (PVOUT) of 1105 MWh per MW

Lazard (2018) has done a comprehensive study on US market on LCOE of renewable and traditional energy technologies. Even without subsidies, utility scale solar is already competitive and even cheaper in some cases than conventional energy sources.

1.3.5. Recycling

As we already mentioned, globally installed PV capacity was approximately 485 GW at the end of 2018, but it is expected that installed capacity will reach to 4.500 GW by the 2050. Particularly high installed capacity rates are expected in China (1.731 GW), India (600GW), The US (600 GW), Japan (350 GW) and Germany (110 GW) (IRENA, 2016).

As the global installed PV capacity increases, so will the volume of decommissioned panels. For an industry which is all about being green and renewable, end of life cycle treatment cannot be ignored by manufacturers, consumers and regulators. Without proper recycling processes and use of recycled materials, the PV industry cannot really be considered green, especially due to exponentially increasing PV waste around globe, unless adequate steps are taken. IRENA (2016) projects global cumulative solar waste for 2 scenarios: the regular loss scenario and the early loss scenario. In the early loss scenario, all the PV panels are recycled at the end of their useful lifetime. In the early loss scenario, the PV panels are recycled earlier, because of malfunctions or other problems. It is projected that global cumulative solar waste will be 60 million tonnes with regular loss scenario or 78 million tonnes with early loss scenario in 2050 (Figure 16).

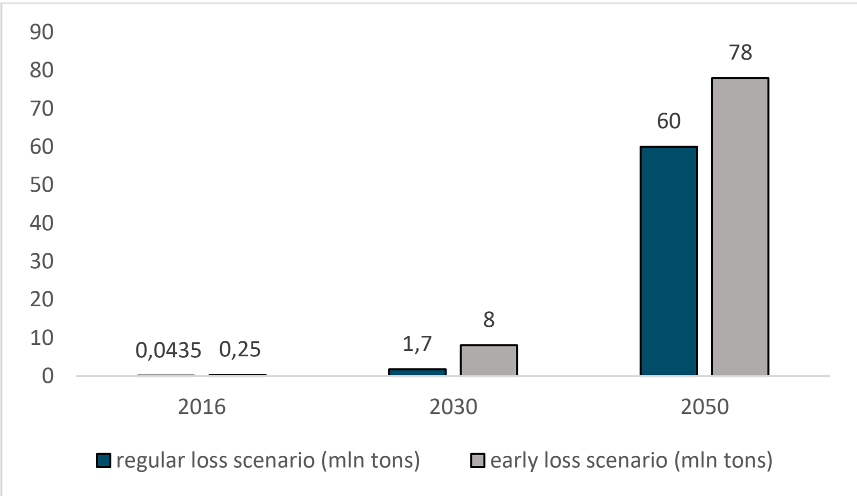

Figure 16: Global PV waste projections in '000.000 tonnes (IRENA, 2(

So far only the European Union has adopted PV specific waste regulations. In the US, China and Japan they are generally classified as general electronic waste and are treated accordingly. EU Waste Electrical and Electronic Equipment (WEEE) Directive requires that all producers supplying PV panels to the European market should finance collecting and recycling end of life PV equipment, regardless of the origin (European Parliment, 2012). By 2030, recycled PV waste might amount to €405 million in value and contain enough material to build 18GW of solar capacity without investing in new raw materials. By 2050, this value could become €13,5 billion, equivalent to 2 billion panels or 630 GW (more capacity than cumulatively installed PV power in 2018) (IRENA, 2016).

PV waste can be classified into different categories such as inert waste, hazardous waste and non-hazardous waste. Sometimes the origin of waste is also taken into consideration such as industrial waste and domestic waste and sometimes by product type, such as e-waste, construction waste and mixed solid waste (IRENA, 2016). In 2017, more than 95% of installed capacity was silicon based (Fraunhofer ISE, 2017). Glass, polymer and aluminium, which can be categorized as non-hazardous waste, make up more than 90% of the mass of these panels. However, some small parts of c-Si panels contain silicon, silver and some small amounts of other elements, such as tin and lead. Together, these materials make up 4% of the mass and they can present recycling difficulties. Thin film panels consist for more than 98% of glass, polymer and aluminium and for 2% of copper and zinc, which is environmentally hazardous. They also use hazardous or semiconductor materials such as indium, gallium, selenium, cadmium, tellurium and lead (appendix 13) Hazardous materials need specific treatment and depending on the jurisdiction, they may fall under specific waste classification (IRENA, 2016).

Responsibility for end-of-life management is generally borne by three main stakeholders: society, consumers and producers. Society is in the form of government controlling and managing operations through taxation. The drawback of this approach is that it may lack competition and innovations might be slow. Another approach is that consumers producing panel waste are responsible for disposal. Consumers might try to reduce costs, therefore limiting the development of sound recycling processes. Consumer approaches are most widely used around the globe. Lastly, producers are responsible based on the extended producer responsibility principle, which means producers are financially and physically responsible for the environmental impact of their products. Although the cost can be passed on to customers through increased prices. The third approach, where producers are responsible, is adopted by EU through WEEE directive. PV Cycle is an example of this scheme. It is a partnership between industry and EU regulators. Under this system, recycling is fully financed by member companies and producers are responsible end of life management, regardless of production locations. Moreover, they must inform users about dedicated collection facilities and about the fact that takeback and recycling are free (IRENA, 2016).

Similar to wind turbines, there are three approaches regarding the recycling of PV panels: reduce, reuse and recycle. The first preference is reduction of materials and increasing efficiency. At the time, strong market growth, scarcity of materials and downward pressure on prices are driving more efficient production techniques, reduction in material use and even substituting existing materials for safer and cheaper ones. The reuse option encompasses repairing and reusing existing panels. As PV panels become more widespread and the maturity of panels increases, a significant second market for PV panels emerges. The least preferable option is recycling. Since there are not much PV panels ending its useful life, recycling is usually done in general recycling plants. That still achieves high material recovery, though some high value materials may not be fully recovered. Solar PV waste recycling has been researched for the last 15 years. In the future, it is expected that solar specific recycling plants will be operational wide scale. In France, water and waste group Veolia opened Europe's first solar specific recycling plant (Reuters, 2018).

Some points also need to be taken account in the future for effective solar waste recycling. Firstly, further damage to the PV panels during dismantling, collection and transport phases needs to be avoided. Furthermore, as many valuable, scarce (indium, tellurium) and hazardous materials (cadmium, lead, selenium) as possible need to be reclaimed. Moreover, recycling-friendly panel designs need to be designed etc. (IRENA, 2016).

Major components of c-Si panels including copper, glass and aluminium can be recovered with more than 85% of mass. However as said before, recovering small amounts of scarce, valuable and hazardous components require specific recycling techniques for solar panels. Recycling thin-film technologies is still in its early stages. However, using current techniques, 90% of glass and 95% of semiconducting material can be recycled (IRENA, 2016). First Solar, which produces thin-film panels have its own recycling facilities in Germany, US and Malaysia (First Solar, n.d.).

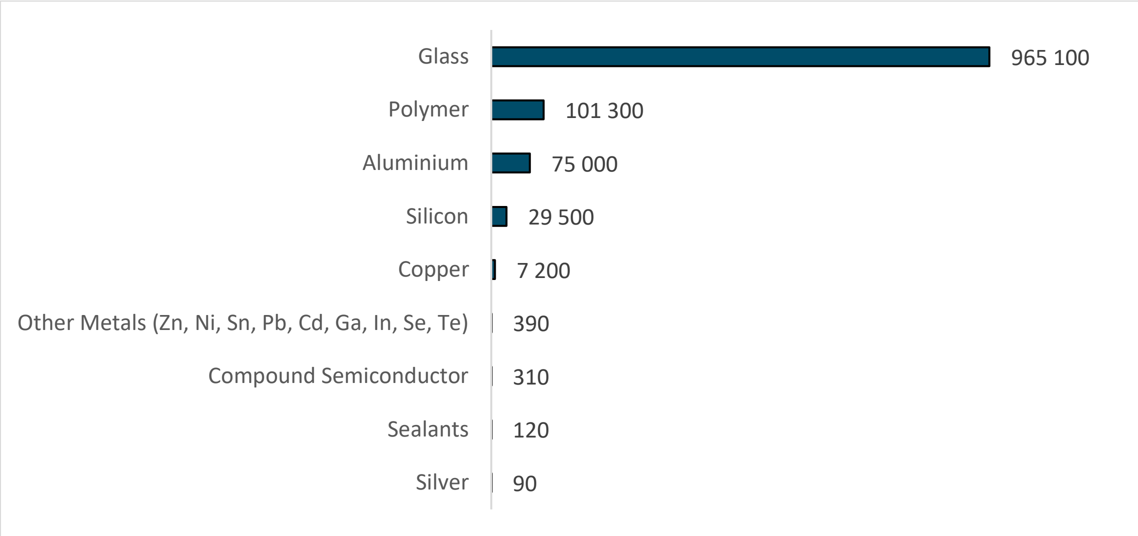

Figure 17: Expected end-of-life recovery rates from PV panel recycling until 2030 (tonnes) (IRENA, 2016)

Under regular-loss scenario the amount of materials is expected to be recovered until 2030 is shown in figure 17. Up to 30,000 tonnes of silicon is expected to be recovered and the value is estimated to be €342 million, assuming silicon prices of €18/kg. The value of silver, of which the expected recovery is 90 tonnes, is estimated to be €45 million, which is enough to make 50 million new panels. Up to 390 tonnes of other materials, including zinc, nickel, gallium, indium, selenium, tellurium and others, are expected to be recovered as well. The value of these metals is €162 million and up to 60 million new panels are expected to be produced.

1.3.6. Carbon emissions

Up to 80 to 95 % carbon emissions of solar panels come from upstream, that is mining of materials and production of panels. While carbon footprint is important, other impacts of PV panel manufacturing such as acidification, eutrophication, abiotic resource consumption and particulate matter emissions are also important. However, 48g of CO_2 per kWh is still 94% and 90% lower than the emissions of coal and combined gas cycle plants respectively.

As a result, lifetime carbon emissions of solar panels are significantly lower than traditional energy technologies such as coal and gas plants. Research done by The Intergovernmental Panel on Climate Change (Schlömer et al., 2014), which is the UN body for assessing science relating the climate change shows that compared to coal and combined cycle gas turbines, utility scale solar produces on average 48 g CO_2/kWh of emissions, compared to coal and combined cycle gas of 820 and 490 g CO_2/kWh respectively (appendix 4).

1.4. Overview LCOE results

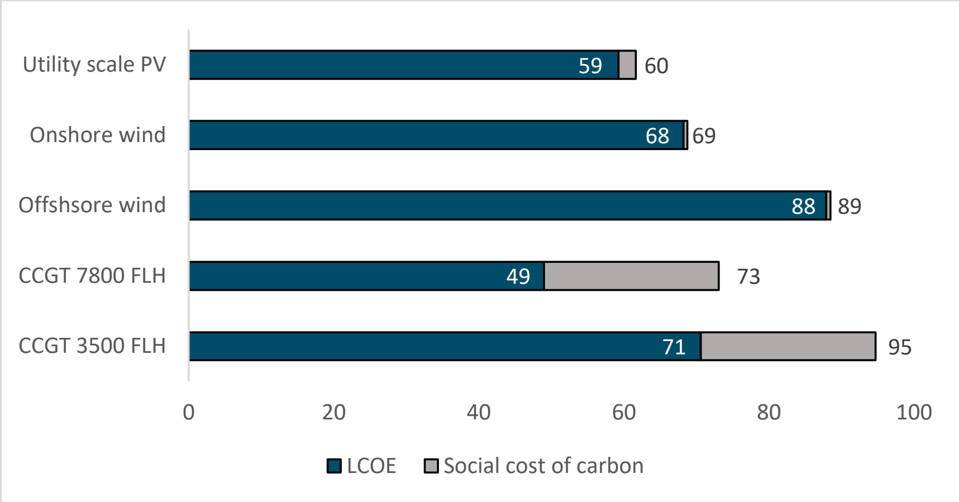

Figure 18: comparison LCOE results, including CCGT and the social cost of carbon

Figure 18 shows an on overview of the LCOE results. Figure 18 clearly demonstrates our finding that utility scale PV is the cheapest renewable energy source, although the gap with onshore wind energy is relatively small. While we already found that the difference in LCOE between on- and offshore wind for Belgium is relatively small compared to other countries, offshore wind is still substantially more expensive than utility scale PV and onshore wind energy.

The calculation of the LCOE for combined cycle gas turbines (CCGT) is added, so renewable energy can be compared fossil fuels. The calculation of the LCOE of CCGT for 3500 full load hours (FLH) is based on data from Fraunhofer ISE (2018), but using our own discount rate. CCGT is mostly used at times when the energy generation through wind and solar is low and during peak hours. According to the study, the amount of FLH for CCGT will likely decrease in the future because more energy will be generated through renewable energy. However, the LCOE is also added for the case when the CCGT is running closer to optimal capacity, at 7800 FLH. This is done, because the same assumption was made for renewable energies. If used at a higher capacity, CCGT is still cheaper than renewable energy. Nonetheless, in practice, it isn't used when there is enough energy produced by other energy sources and the LCOE is higher than utility scale PV and onshore wind. Given the fact that CCGT are used at times when there is low energy supply or high energy demand, the price of the generated energy will also be higher, and this increases profits for CCGT.

In figure 18, the social cost of carbon is added, based on the mean estimate for the social cost of carbon of €49 per ton of carbon emissions. This is the mean of the meta-analysis of Wang, Dengh, Zhou & Yu (2019). It is also based on the carbon emissions per energy technology of IPCC (Schlömer et al., 2014). The social cost of carbon could be considered as the cost to society, for example through climate change. When this is added, utility scale PV and onshore wind energy are the cheapest solutions.

1.5. Other LCOE studies

Most LCOE are done in the United States. Because of this, they are mostly reported in USD. While we usually changed these numbers to EUR, they are not changed in this part.

Bloomberg new energy finance

Bloomberg new energy finance (BNEF) shows that the LCOE of onshore wind and solar PV without tracking systems went down 18% in 2018 compared a year before. The LCOE of offshore wind went down 5% compared to 2017. They found that, in 2018, the LCOE of onshore wind, solar PV and offshore wind was 50 €/MWh, 64 €/MWh and 107 €/MWh respectively. Also, between 2010 and 2018, the price of lithium-ion batteries went down 70% which makes economic viability of building coal and gas power plants diminish further and it is expected that it will go down further 66% from 2018 levels until 2030. They also predict that by 2050, solar and wind technology combined will provide 50% of electricity generation worldwide and role of fossil fuels will go down just to 29% compared to 63% today (Bloomberg New Energy Finance, 2018).

IRENA

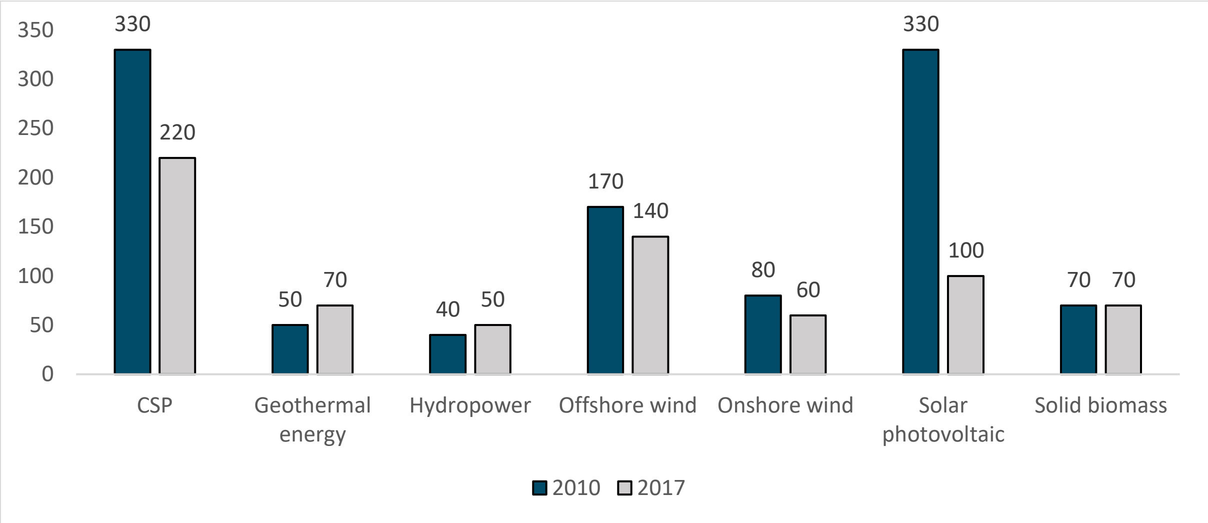

Figure 19: LCOE (USD/MWh) of renewable energy technologies (2010-2017) (IRENA, 2017)

IRENA data show that the price of renewable energy sources, specifically wind and solar, went down considerably between 2010 and 2017. Figure 19 shows LCOE changes of major renewable energy sources (IRENA, 2017). IRENA data reported very wide ranges of minimum and maximum values, which can be found in appendix 11.

Lazard

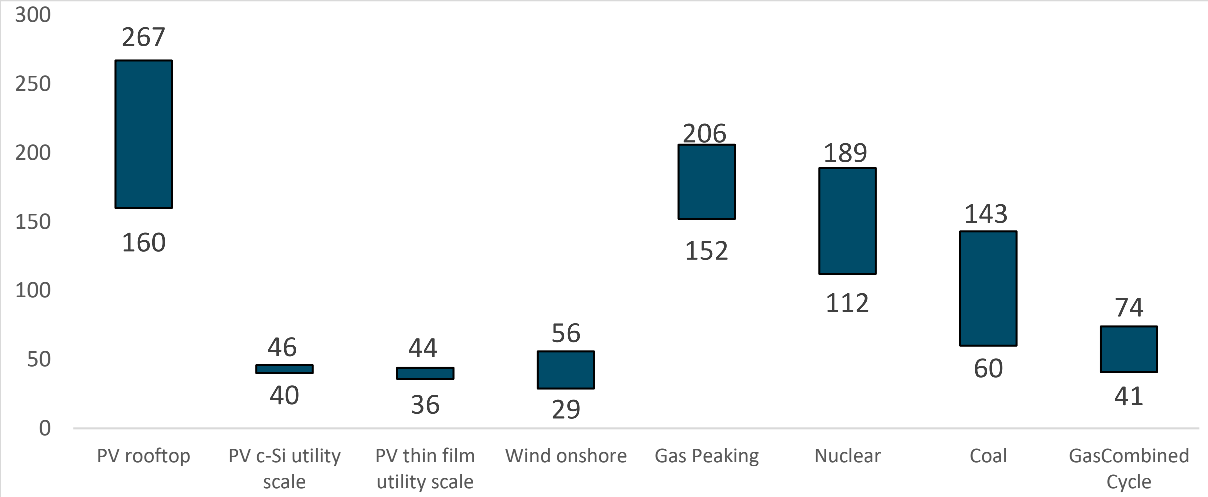

Figure 20: LCOE of renewable and conventional energy technologies for US (Lazard, 2018)

From Figure 20, it can be seen that utility scale solar crystalline and thin film, as well as onshore wind technologies are already the cheapest sources of energy among other renewable and conventional sources in US according to Lazard (2018). Solar PV crystalline and thin film LCOE's range from 40-46 USD/MWh and 36-44 USD/MWH respectively. Onshore wind LCOE is 29-56 MWh/KWH. They are already cheaper than gas peaking, nuclear and coal in USA. The majority of new energy investments worldwide go to renewables with solar and wind leading the way. The values of Lazard are lower, for example because of better onshore wind conditions in the US.

Historical development of LCOE from 2009 to 2018 is shown (appendix 12) in Lazard's US analysis. Utility scale crystalline panel LCOE is reduced from 359 USD/MWh to 43 USD/MWh from 2009 to 2018 which is approximately 88% decrease over the 9-year period. Similarly, the LCOE of wind energy went down 69% from 135 USD/MWh to 42 USD/MWh. In contrast, Nuclear LCOE increased from 123 to 151 USD/MWh, 22.8% increase during that period.

1.6. Energy storage

Next to using gas turbines at times when demand exceeds supply, electricity can also be stored by converting it to another form, such as potential, kinetic and chemical energy. Electricity storage technologies consist of several types of storage mediums. These are chemical batteries, flow batteries, fuel cells, flywheels, superconducting magnetic energy storage, super capacitors, compressed air energy storage and pumped hydro. For example, pumped hydro storage systems store energy in the form of water in upper reservoir to be used later (Nirmal-Kumar & Garimella, 2010).

In this part we will focus on electric batteries, more specifically on lithium-ion batteries. In general, there are several types of electric batteries such as lead-acid, nickel-cadmium (NiCd), nickel metal hydride (NiMH), lithium-ion (Li-ion) etc. Since li-ion are the most widely used and promising technology for the future, the levelized cost of storage in terms of li-ion batteries will be investigated.

Comello & Reichelstein (2019) analyzed the levelized cost of energy storage (LCOES) in terms of the li-ion batteries for behind-the-meter residential application in Germany and California in the US. The levelized cost of energy storage (LCOES) metric is considered to be the minimum price per kWh stored that an investor requires over the entire lifetime of the storage facility. As solar and wind energy usage increases, issues like intermittency and dispatchability gain importance. Batteries store energy when it is not needed, and that energy can be used later. Residential behind-the-meter storage's economic benefits rise from the difference between retail prices of electricity and the overall tariff that is obtained for surplus energy generated by PV, but not self-consumed. As an example, during the times that PV do not generate electricity, energy needs should be purchased from the grid at retail prices if batteries are not available. If the average lifetime cost of using per kWh of battery capacity is less than average lifetime retail prices, it is economically viable to install a battery for behind-the-meter residential needs.

In Germany, feed-in tariffs, which is the price of electricity sold back to grid, is 0,12 €/kWh and the retail prices of electricity is on average 0,30 €/kWh. That creates a substantial price premium 0,18 €/kWh. LCOES are calculated as 0,085 €/kWh, which is less than difference between retail prices and feed-in tariffs (net price paid for purchasing electricity from the grid). That means battery installation for residential solar is economically viable for Germany (Comello & Reichelstein, 2019).

In California, the State introduced a net metering system. It means that residents can sell excess capacity to the grid and buy it back when necessary at the same rate, which effectively means free storage. However, the State also introduced tax cuts and other incentives such as non-bypassable charge, which means any electricity sold back to grid will be credited at basic retail rate for Solar PV storage applications. LCOES amounts to 0.0054 €/kWh for California and the net price paid to the grid is regulated in a way that incentivize battery installations (Comello & Reichelstein, 2019).

A BNEF study on batteries shows that, as solar and wind becomes the cheapest source of bulk generation, the importance of batteries increases. It is expected that, by 2050, 1291 GW of battery power will be added and cost of a battery pack for stationary applications will be 64 EUR/kWh (BNEF, 2018).

Lazard has made a study on levelized cost of storage (LCOS) of different use cases, divided into two groups (In-front-of-the-meter and behind-the-meter) and battery technologies. In-front-of-the-meter use cases are wholesale, transmission and distribution and utility scale (PV+Storage). Behind-the-meter cases are commercial & Industrial (standalone), Commercial & Industrial (PV storage) and residential (PV storage). Battery technologies assessed are Lithium-ion, Flow Battery-Vanadium and Flow Battery-Zinc Bromide for in-front-of-the-meter applications and lithium-ion, Lead Acid and Advanced Lead (Lead Carbon) for behind-the-meter applications (appendix 14) (Lazard, 2018).

As a result, the levelized cost of storage (LCOS) is obtained for different use cases and battery specifications. As appendix 15 shows, the lowest LCOS is obtained by utility scale solar PV with storage using li-ion batteries with a range of 108-140 USD/MWh. Lazard's LCOS analysis is constructed by creating an energy storage model representing an illustrative project for each relevant technology and use cases and solving for the USD/MWh figure that results in a levered IRR equal to the assumed cost of equity, for example 12% (Lazard, 2018). This LCOS expressed in USD/MWh also represents minimum price for stored electricity in order to break even. Based on different use cases and project specifics, LCOS represents the minimum retail price that electricity needs to be sold to make it economically viable.

After setting LCOS for different use cases and battery types, revenue streams are identified (appendix 16). For example, energy arbitrage, which means buying electricity when it is cheaper and selling at peak hours when the prices are higher. Another example would be backup power which is using own residential storage when the grid is down or resource adequacy for utility scale PV with battery which means to provide electricity at peak loading at the regions with limited generation and transmission capacity.

The next step after identification of these revenue streams for selected use cases, IRR and percentage of IRR for different revenue streams are identified for various select regions in US and internationally (appendix 17, 18). For example, utility scale PV with storage in West Texas has IRR of 8,8%, with 66% of it coming from energy arbitrage. Another example is utility scale PV with storage in Australia has an IRR of 8.7% with 73.8% of it being energy arbitrage.

For different battery technologies, future capital cost of technologies and some trends for the near future identified (20).

Battery Technologies	CAGR	Price reduction by 2022
Lithium-ion	8%	28%
Flow Battery-Vandanium	11%	38%
Flow Battery-Zinc Bromide	14%	45%
Lead Acid	3%	13%
Advanced Lead (Lead Carbon)	4%	17%

Table 12: Battery technologies and reductions in the capital cost until 2022 (Lazard, 2018)

2. Rare-earth elements and other critical elements

In this part, we first investigate the reserves and production of rare-earth elements, as well as their environmental impact. Certain wind turbine models using permanent magnets contain rare-earth elements. We next look at the impact of this on wind turbines. Solar panels don't include rare-earth elements, but other critical elements which are also investigated in this part.

2.1. Rare-earth elements

The rare-earth elements (REE) are a group of 17 elements. These 17 rare-earth elements include the 15 lanthanides, with atomic numbers 57-71, as well as Scandium and Yttrium, with atomic numbers 21 and 39 respectively. The lanthanides are often divided into 2 groups: heavy rare-earth elements (HREE) and light rare-earth elements (LREE). Yttrium and Scandium are often considered as a heavy rare-earth element, since they have similar characteristics. Appendix 19 provides an overview of all rare-earth elements, with the atomic number, symbol and crustal abundance (Van Gosen, Verplanck, Long, Gambogi and Seal, 2014).

REEs have particular electronic structures, which offer unique properties (Navarro & Zhao, 2014). Because of these unique properties, they are often used in high-tech consumer goods, such as smartphones, flat-screen monitors and televisions, but also used in large quantities in technologies crucial for the energy transition, such as electric and hybrid vehicles, wind turbines and solar panels. They are also difficult to substitute, because of the unique properties. Furthermore, rare-earth elements are also needed in certain defence technologies, which also make them strategically important for countries. For example, they are used in night vision googles, GPS equipment, communication equipment and other critical defence technologies. Substitutions exist for these technologies. However, they are generally more expensive and less effective, which diminishes military superiority (King, sd).

The useful and unique properties of these REEs caused demand to increase drastically during the last years (King, sd). Since REEs are important for evolving technologies, it is suggested that high potential exists for disruptive demand (Van Gosen et al., 2014). The European commission also identifies both light and heavy REEs as critical raw materials. Certain materials are considered to be critical based on 2 criteria: when they are of high economic importance and when the risk associated with the supply is high. Of all materials, LREEs and HREEs were considered as the 2 material categories with the highest supply risk (European Commission, 2017).

Rare-earth elements are called rare because at the moment they were discovered, in the 18th and 19th century, they were relatively rare compared to other elements which met the criteria for being defined as 'earth's' (Van Gosen et al., 2014). In practice, they are not necessarily rare. For example, cerium is ranked 25th of the 78 common elements in crustal abundance (US Geological Survey, sd). Appendix 19 gives an overview of the crustal abundance from each rare-earth element. None of these elements is less common than gold and silver, except for promethium. Promethium is radioactive and only 500-600 grams of it naturally occur in the earth's crust and, as a consequence, it is very rare. Of the other elements, none of them are less common than gold or silver and Cerium is, with 66,5 parts per million, even more common than copper.

While most REEs are relatively abundant in the earth's crust, they are mostly found in low concentrations and rarely in economic concentrations (Van Gosen et al., 2014). REEs don't exist individually like gold or copper but are located in minerals. Even for the economically minable deposits, the concentrations are still low (Pitron, 2018) and recovery of REE from the ores can be complex and costly (Van Gosen et al., 2014). Different REEs are typically found in the same ore deposits (Wyomong State Geological Survey, sd) and as a result, they are most often co-mined. REEs are sometimes also a by-product of the mining of other elements. For example, in the Bayan Obo mine, one of the biggest reserves of REEs, the REEs are by-products of iron ore extraction. For every 350kg of iron ore, 60kg of REE are obtained and 1,3 kg of niobium (Navarro & Zhao, 2014).

The REEs need to be processed in order to separate REEs from the ore. The process is different in processing according to the mined minerals, as well as the specific refinement facility. In general, the REEs are extracted using a combination of hydro-metallurgical techniques and acid baths, using acids like sulphur and nitrogen (Bontron, 2012). It takes tens of repetitive operations to achieve a concentration of REE close to 100%. Next to sulphur and nitrogen, 200 cubic metres of water are required for each tonne of REE. Meanwhile, these 200 cubic metres of water become filled with acid and heavy metals. Furthermore, the ores often contain radioactive elements such as thorium. This results in massive amounts of toxic and radioactive waste (Pitron, 2018; Maughan, 2015).

2.1.1. Reserves

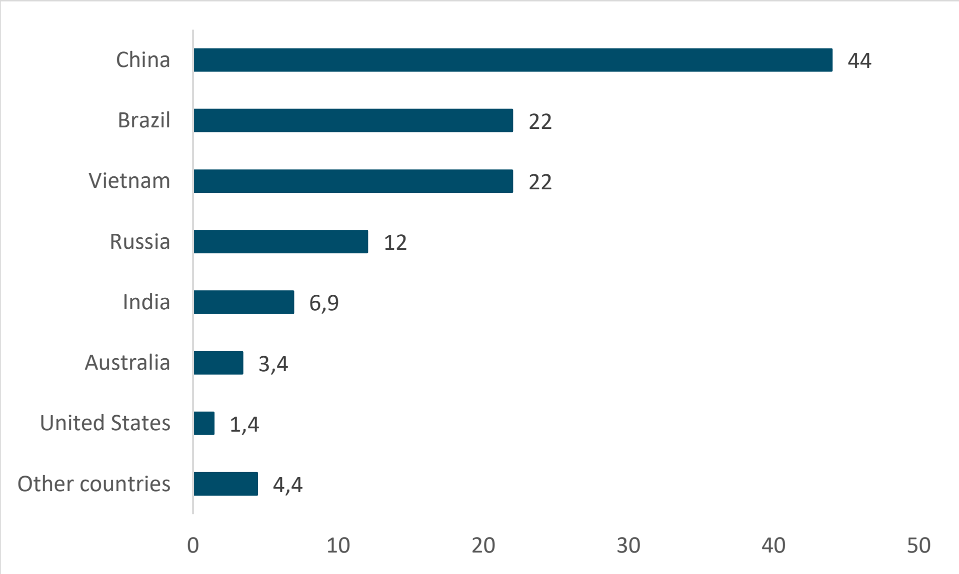

Figure 21: Reserves of REEs per country in '000.000 tonne (US Geological Survey, 2019)

Figure 21 gives an overview of the known REE reserves in the world. It is clear that by far the largest reserves are located in China, followed by Brazil and Vietnam. It is remarkable that the united states are only ranked 7th, while most REE production came from the United States before the closing of the Mountain Pass mine in 1990. However, these reserves of 1,4 million tonnes is still 8 times the REE production of 2019, which was 170.000 tonnes. The total reserves in the world are estimated to be 120 million tonnes, which is over 700 times the production of 2018. The reserves reported in figure 21 are defined as dynamic and it are the reserves could be extracted economically at the point of determination (US Geological Survey, 2019). There are also other reserves, which are found in less economic concentrations. However, when the amount of reserves decreases, the price increases, resulting in more reserves being economically viable to mine and more reserves will be discovered due to increased exploration. Appendix 20 and 21 also provide an overview of the reserves per country, including the reserves as a percentage of the total reserves in the world.

Reserves are mostly found in bastnaesite (Northern China, United states), monazite (Northern China), Xenotime (Malaysia) and Ion-absorption clays (Southern China). Bastnaesite is mined most commonly, and xenotime isn't mined at a significant level (Navarro & Zhao, 2014). Most REEs are mined using open pit mining, and, in heap leaching or in situ leaching for ion-absorption clays in the South of China (Yang et al., 2013).

2.1.2. Historical production

Appendix 22 provides an overview of a timeline with the amount of REE produced in the most important countries. Before 1965, most REE were mined in South-Africa, India and Brazil, but the amounts mined were relatively small, at less than 10.000 tonnes per year. After 1965, demand increased due to new technologies such as colour television. Because of this, the United States became the biggest producer and the global amount produced increased to 50.000 tonnes. The REEs were mainly provided by the Mountain Pass mine in California (Pitron, 2018) and it accounted for 70% of the world's supply in the 1970's and 1980's (Navarro & Zhao, 2014).

By the 1990's, Mylocorp, the owner of the Mountain Pass mine, had a lot of issues with wastewater leaking from its piping system. This caused environmental concerns and the government introduced more stringent regulations. This required substantial investments from Mylocorp, for example for new infrastructure in the mine. At the same time, China was booming economically and seized the opportunity to increase its REE production. The combination of having the biggest reserves in the world, lenient environmental regulation and low labour costs, allowed China to produce REEs at a low price. These factors led to the closure of the Mountain Pass mine in 2002, but mining activities were already stopped in the 1990's (Pitron, 2018) and allowed chine to became dominant in the REE market. In 1993, China and the United States both produced 33% of REE, but in 2010, 95% of REE came from China (American geosciences institute, sd).

In 2008, after almost 2 decades of dominance in the market by China, China started to implement production quota (Navarro & Zhao, 2014). In 2010, there was a dispute between China and Japan in the sea around the Senkaku Islands. This resulted in an embargo, causing China to stop exporting REEs to Japan (Pitron, 2018). In the same year, China reduced its export quota of REE by 40%. Prices increased in the years preceding 2010 and during 2010 with more than 500% (King, sd). This caused demand to exceed supply, leading to concerns in other countries, particularly the ones dependent on high-tech such as the United states and members of the European union. However, China realised they were also dependent on other countries for its trade and increased its exports. They also ended its embargo with japan (Pitron, 2018). The WTO also ruled against china in 2014 and China wasn't allowed to use export quota anymore.

Nonetheless, other countries wanted to limit their dependence on China, and, as a result, action was taken in order to discover deposits of REEs and bring them into supply. The European Commission's Raw Material Supply Group made an action plan to improve access to supply which increase market accessibility, investments in R&D and recycling programs. Japanese firms and the Japanese government also looked for mines in Asia, Africa and the Americas and the US DOE also released Critical Materials Strategies to reduce dependence on China (Navarro & Zhao, 2014).

The effort of other countries to take control of REEs themselves is illustrated by China's share in REE production decreasing from 80% in 2017 to 71% in 2018 (US Geological Survey, 2019). The reason for this wasn't that China produced less because its absolute production increased with 14%. The production from other countries simply increased relatively more, so their market share of China decreased (appendix 20, 21).

2.1.3. Current production

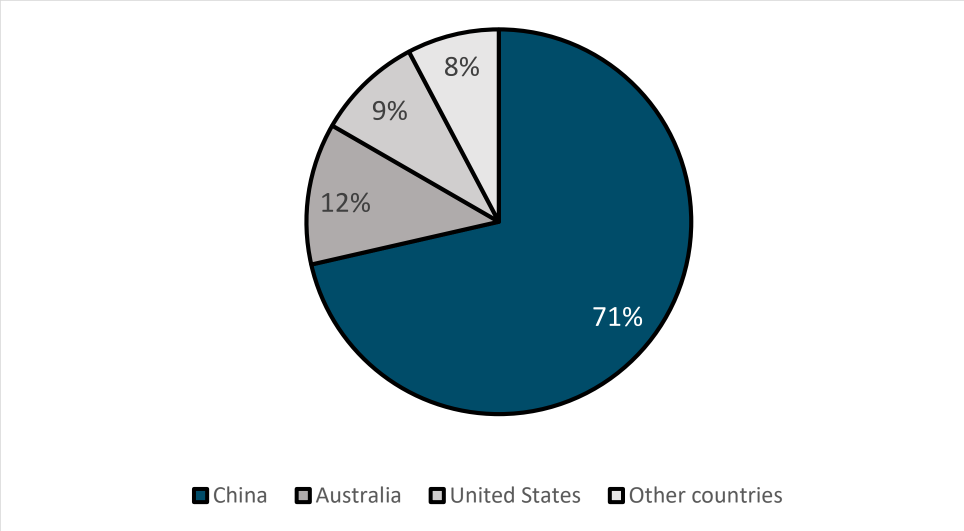

Figure 22: REE production in 2018 (US Geological Survey, 2019)

Figure 22 provides an overview of current production per country. Clearly, China is still the dominant player, but competition has increased, particularly from Australia and the United States. This is somewhat remarkable, given that the United States and Australia are only ranked 6th and 7th in REE reserves (US Geological Survey, 2019). Australia began opening mines in 2011 and the united states reopened the mountain pass mine in 2012. Other countries also increased production since 2018. In 2018, there was, next to the countries mentioned in figure 22, also significant production in: Burma (Myanmar), Russia, India, Brazil, Burundi, Vietnam and Malaysia (Appendix 20, 21).

Figure 22 might underestimate the share of certain countries in total production. It is mentioned that there is illegal production in China which is not included in figure 22 (US Geological Survey, 2019). Between 2006 and 2010, the gap between China's export quota and the production reported by the US geological survey was as big as 30%. Furthermore, there are some small illegal mines in Brazil, Thailand and Vietnam (GEUS, 2017).

China

In China, the market is dominated by 6 companies: 'The Big Six'. These 6 companies are the result of consolidation in the industry in 2013, encouraged by the Chinese government (GEUS, 2017). They are state-owned and vertically integrated. These companies are China Minmetals, Chinalco (Aluminum Corporation of China Limited), China Northern Rare Earth Group (Baotou Steel, Baogang group), Xiamen Tungsten, Ganzhou Qiandong Rare Earth Group and Guangdong Rising Nonferrous Metal (Mancheri, 2015). It also appears like some of them are horizontally integrated. For example, Chinalco is China's biggest aluminium producer. According to Mancheri (2015), there are 87 recognised rare-earth enterprises, but numerous advantages were given to 'The Big Six' through legislation and financing. They are licensed to take over small companies or illegal mines and 90% of production quotas are allocated to these 6 companies (Mancheri, 2015). While it is true that the companies are state-owned, many of these companies are also partially publicly traded (Bloomberg).

The fact that China consolidates these enterprises and makes them state-owned, is a sign that it considers the REE industry as more strategic than in the past. There is also the existence of export quotas and export tariffs. This way, Chinese enterprises further down the supply chain have cheaper access to REEs compared to foreign companies, and as a result, China can further develop its high-tech industry and have control from mining to end-product (Mancheri, 2015). From 1990 until 2009, China's domestic consumption of REEs for high-value added product manufacturing has increased annually with 13% (Tu, 2010).

The production in the North of china (e.g. Bayan Obo mine) mainly consists of LREEs, whereas the share of HREEs is higher for the ion-absorption clays in the Southern provinces (Pitron, 2018). Furthermore, the production of illegal REEs could be well over 30%, but there are almost no data available about this (Liu, 2016).

Rest of world

In the United States, the production is provided by MP Materials, which is the owner and operator of the Mountain Pass Mine (MP Materials, sd). They bought the mine from Mylocorp for $20,5 million in 2017 (Topf, 2017). This was after Mylocorp filed for bankruptcy in 2015 because high debt, which they couldn't pay back because of low REE prices. Their debt was $1,7 billion, mainly caused by investments in their in facilities. Mylocorp bought the mine in 2010 and reopened it after it seized operation in the 1990's (Topf, 2017). The REEs are both mined and processed at the mining site (MP Materials, sd). Some REEs are separated on the site, but not all of them. For example, some rare-earth concentrate is hipped to Estonia for processing (GEUS, 2017).

In Australia, Lynas Corporation operates the Mount Weld Mine in Australia and the concentrate mined is shipped to the Lynas Advanced Materials Plant (LAMP) in Malaysia for processing. The LAMP facility has a capacity of 22.000 tonnes per year and is located in the city of Gebeng, which has a population of 1,5 million people. Most of the REEs produced is praseodymium/neodymium, which is often used in wind turbines. Both the mine and the processing plant are owned by Lynas corporation, a stock listed company headquartered in Malaysia, with a market cap of €1,08 billion. The reasons for Lynas choosing Malaysia as the location for the processing site, are low labour and construction costs (Kuan, Ghorbani & Saw, 2016). Liu (2016) mentions that the main reason is more lax environmental regulation in Malaysia. Moreover, Japanese corporations have agreements with Lynas to secure supply. This is manifested by three Japanese firms agreeing a 10-year supply of 8.500 tonnes per year of REEs in 2011 (Thompson, 2011).

Rest of world

There are also various smaller mines around the world. For example, in Burundi, there is Rainbow Rare Earths company, which is listed on the London Stock Exchange. They stared producing since the end of 2017 and they operate the Gakara Rare Earth Project in Burundi. According to the US Geological Survey (sd.), the annual production from Burundi in 2018 was 1.000 tonnes. However, they agreed the sale of 5.000 tonnes of REE concentrate per year with ThyssenKrupp Raw Materials for a period of 10 years (Rainbow Rare Earths, sd).

Future supply

Clearly, there is a trend for increased REE production outside China to reduce dependency. Promising projects in Europe include Norra Kärr in Sweden, Kvanefjeld and Kringlerne in Greenland, Fen in Norway and Aksu Diamas in Turkey. It is estimated that a combination of supply from Norra Kärr, Kvanefjeld and Kringlerne would be sufficient to ensure supply in Europe for decades (Balomenos et al., 2017). According to Schreiber, Marx, Zapp, Hake, Voßenkaul and Bernd (2016), important advantages of a mine in Europe are better environmental standards and reduced dependency on Chinese supply. The drawbacks of the European mine are higher cost and less social acceptance of mining activities in Europe.

Outside Europe, GEUS (2017) identifies 6 other promising, advanced-stage projects. These include Dubbo and Nolands in Australia, Kipawa, Strange Lake and Nechalacho in Canada and Steenkampskraal in South Africa. These projects are all considered developed through a feasibility study and could be operational from 2025 onwards. However, not all of them will reach the operation stage, since demand would exceed supply (GEUS, 2017).

García et al. (2017) estimated the financial returns for 5 REE projects outside China, and all of them reached higher returns than the cost of capital. With 43,4%, the Kvanefjeld project in Greenland had the highest internal rate of return. However, this is highly dependent of forecasted prices and they concluded that more conservative assumptions should be used to achieve investors' confidence. According to GEUS (2017), economic performance might not be sufficient and the REE market in Europe would need substantial political support.

The trend of increase in production outside China is also manifested by the emergence of companies like Ucore Rare Metal. This is a Canadian company which doesn't produce any REEs yet, but already is publicly listed and has a market cap of €44m. However, they have full ownership of the Bokan-Dotson Ridge mine, of which the ores have the highest concentration of HREES in the United States (Ucore, sd). Another example is a Canadian company will start producing 3.000 tonnes per year from 2020 onwards, based on a mine in Malawy in Africa (Jamasmie, 2017).

2.1.4. Environmental impact

China

Given that China is the biggest producer of REEs, most studies on the environmental impact are based on Chinese production. In China, it appears like only the most basic ecological and sanitary standards are maintained. The water used in the process, contains acids and heavy metals after extraction of REEs, only rarely gets purified before it is released back into rivers, the soil or artificial lakes. As a result, the REE industry is considered to be one of the most polluting in China (Pitron, 2018). The environmental damage was also mentioned as one of the reasons for China to reduce its exports in 2010 (Ali, 2014).

Northern China

Figure 23: toxic waste dumped into artificial lake in Baotou (Maughan, 2015)

An example of environmental damage in China is the situation in Baotou. This city is located in the North of China, north-west of Beijing and is located close to the Bayan Obo mine, which is said to hold the biggest REE deposit in the world (Song et al., 2018). The minerals mined at Bayan Obo are brought to the refineries in Baotou for processing. All toxic waste from the refinery process is dumped into a giant artificial lake next to the city (figure 23), at a rate of 10 million tonnes per year (Kaiman, 2014). No fish or algae are able to survive in the lake and the water is reported to seep into groundwater (Maughan, 2015; Kaiman, 2014). Liu (2016) mentions that the pond is non-permeation proof and that it is still expanding, and threatens to be a disaster for the yellow river, 10 kilometres away from the pond. There are also reports of failing crops in nearby villages, which caused farmers to move away. The air in the area contained sulphuric acids and coal dust, which the citizens inhaled. The coal dust was caused by fossil power plants, of which the energy was used to extract the REE (Bontron, 2012). According to Kuan et al. (2016), China committed $600 million to clean up the environmental damage.

Academic literature quantifies these issues and provides information about the environmental impact of this industry. In terms of global warming, research found 32,29kg CO_2 equivalent per kg LREE and 34,49kg CO_2 equivalent per kg for HREEs (Koltun & Tharamurajah, 2014). In perspective, the values for iron ore, copper concentrate and gold are 0,0119; 0,63 and 29.820 kg CO_2 per kg. However, global warming appears to have a low environmental impact relative to other issues. According to Schreiber et al. (2016), global warming makes up less than 5% of the total environmental impact at Bayan Obo and the biggest impact is caused by human toxicity and aquatic ecotoxicity. Other important issues are particulate matter and eutrophication[4] of fresh water. Koltun & Tharumarajah (2014) assessed the life cycle impact of REE with the Bayan Obo mine regarding global warming. The biggest environmental impact is incurred during the processing stage.

Southern China

In the South of China, there is also significant REE production. While in the North of China, mostly LREEs are mined, in the South of China, HREEs are more common (Pitron, 2018).

In the South of China, the REEs are extracted from ion-absorption clays using. While these clays only account for 2,9% of China's REE reserves, they accounted for 35% of production in 2009 (Yang et al., 2013). In the past, this was mainly done through surface mining and heap leaching. Heap leaching means that all the clays are taken out of the ground and put onto a heap. Next, a leaching solution is applied to extract the REEs (Carlson & Le Capitaine, sd). It is estimated that, for 1 tonne of REO, 300m² of vegetation and topsoil are removed and 2000 tonnes of tailings are disposed in valleys and streams, as well as 1000 tonnes of wastewater containing ammonium sulphate and heavy metals (Yang et al., 2013). This threatens the safety of drinking water of the Dongjiang river and the Ganjiang river, which are a source of drinking water for millions of people (Liu, 2016). REE mining also caused changes in topography and resulted in more flooding and other disasters. According to Yang et al. (2013), the costs to restore the land are slightly higher than the turnover of REEs in the region and almost 20 times higher than the profit from REE mining. They also mention that heap leaching has led to 191 million tonnes of tailings and 153 km² of destroyed forests.

[4] Eutrophication happens when minerals leak into water, which could cause excessive growth of algae and depletion of the oxygen in the water.

The alternative way of mining of these ion-absorption clays is done through in situ leaching. During this process, holes are first drilled into the ground and subsequently, a leaching solution, containing ammonium sulphate, is poured into the ground. Finally, the solution and the dissolved ore could be pumped out to the surface. The advantage of this is that it only requires clearing of vegetation and removal of the topsoil at the location of the holes. However, still 33% of vegetation needs to be cleared and 7000m³ of slurry is produced per hectare (Yang et al., 2013). There is almost no radioactivity in the clays (Yang et al., 2013), but there are other polluting factors, such as groundwater pollution, landslides and mine collapses. This is manifested by increased concentrations of sulphate and other pollutants, which were found in groundwater (Navarro & Zhao, 2014). The Chinese government encourages use of in-situ leaching to reduce environmental impact (Li and Yang, 2016).

Vahidi, Navarro & Zhao (2016) compared the environmental impact of in situ leaching from ion-absorption clays to open pit mining of bastnaesite and monazite minerals (e.g. Bayan Obo). They found similar results regarding the impact of global warming. However, in situ leaching has a smaller impact regarding acidification but a higher impact in the category of eutrophication.

Black market

There is also a substantial black market in China. This causes even more pollution than legal mining, given that illegal miners do not bother with regulations. They also use basic technologies which are less efficient and discharge of toxic waste without treatment (Liu, 2016).

In general, China has increased its environmental regulation and driven up the amount of inspections since 2010 (Liu, 2016). In 2015, the environmental regulation increased further, and companies could lose licenses if the rules are violated (Mancheri, 2015). The rare-earth industry development and implementation plan was released in 2016. This is a 5-year plan and in this plan, it is mentioned that China wants to increase the percentage of rare-earth companies who comply with the environmental legislation from 30% to 90% (Peak Resources, 2016).

Australia and Malaysia

As already mentioned, Lynas mines ores in the Mount Welt mine in Australia and ships this to the LAMP-facility in Malaysia. While there was some critique to off-shoring the processing to Malaysia in order to reduce environmental damage in Australia, the REE seem to be processed in a more environmental-friendly way compared to China. For example, the wastewater and spent chemicals go through wastewater treatment facility before discharge (Ali, 2014).

Kuan et al. (2016) review the processing of REEs in Malaysia. In the past, there were 2 companies who processed REEs and operations ended in 1992 because of environmental problems, such as vast amounts of radioactive waste. Mitsubishi Chemical Company operated in Malaysia, and they failed to dispose of the waste safely. This caused birth defects and leukaemia among the local population. There was legal action against the company, resulting in Mitsubishi having to pay settlements, as well as a clean-up cost. The clean-up cost the company $100 million. The problems in the past caused resistance by the local community when plans were made to open the LAMP facility. However, in 2012, first shipments to the LAMP facility in Malaysia took place.

Schmidt (2013) studies the environmental impact of the LAMP facility, on behalf of the NGO "Save Malaysia, Stop Lynas" (Save Malaysia Stop Lynas, 2012). He studied the emissions of radon, sulfuric acid and dust, as well as discharges via the water pathway. Firstly, emissions are emitted during the cracking stage and this cannot be filtered. It was found that these emissions could be considered negligible. For the emissions of sulphuric acid and dust, the gas stream passes a waste gas treatment system and the sulfuric acid and dust are removed. However, this is not done for all gases in the facility. The sulfuric acid emissions were found to be too high by a factor of at least 2. Most concerns were regarding the water pathway. They mention that the water used is neutralized, sludges are removed, the water is collected in a pond, then diluted and discharged in the river after monitoring. After 3 km, the water enters the sea. However, they mention that there is insufficient transparency regarding by-products and salt. Finally, then have concerns regarding storage of the waste, but it appears like the company has already solved this issue through the residue management plan (Lynas, sd). Finally, there is issue that the plant is located in a location where there is risk of flooding during the monsoon season and this could be detrimental for the environment.

Both Water leached Purification Residue (WLP) and Neutralization Underflow (NUF) are wastes produced in the LAMP facility. For this waste, Lynas (sd) has a residue management plan. According to the principle of cradle to cradle, they mention that it is preferable to reuse the waste. WLP is radioactive, but it is classified as very low-level radioactive material. For example, the level of radioactivity is similar to certain materials used in UK roads. NUF is not radioactive and rich in magnesium. According to Lynas, these materials could be used as materials to create a soil conditioner. Kuan et al. (2016) point out that thorium could potentially be extracted from WLP and be used as nuclear fuel. However, if the waste could not be reused, Lynas (sd) also assured a permanent disposal facility (Lynas, sd). The tailings of the Mount Weld mine are also stored in storage facilities. The gangue minerals (waste) are chemically stable (Lynas, sd).

In conclusion, it appears like the Lynas facilities are more environmentally friendly compared to Chinese facilities. Effort is taken to limit the impact on the environment and there appears to be more transparency. However, there is still a negative impact, due to certain emissions in the air and water, as well as the production of waste which needs to be stored. Next to the negative impacts, the Lynas facility creates (skilled) jobs (Kuan et al., 2016) and contributes to the economic development of Malaysia.

United States

In the Mountain Pass mine in the United States, the toxic waste was carried to an evaporation system. However, as already mentioned, the mine was closed because of leakages in the piping system, which caused the toxic waste to be leaked in the desert. Since the reopening of the mine, the wastewater system was changed and it is now managed much closer to the mine, through a new chlor-alkali plant to recycle the wastewater (Mining Technology, sd). Kuan et al. (2016) confirm that the Mountain Pass mine has become more environmentally friendly since the reopening of the mine in 2012, thanks to the new management. It is mentioned that environmental regulators revealed general satisfaction with the processing at the site.

On the company’s website, they mention that more than €1.35 billion has been invested in facilities at the Mountain Pass mine since 2010. This was to ensure compliance with the environmental standards, which the company considers to be the most stringent environmental standards in the world. The company mentions that their key innovation regarding sustainability is that they don’t use a wet tailings pond. Through their processing plant, dry waste is generated and stored at a ‘lined impound’ and they recycle the water in the production process (MP Materials, sd).

Europe

Schreiber et al. (2016) assessed the (hypothetical) environmental impact of both the, until now unopened, Norra Kärr mine in Sweden and the real environmental impact of the Bayan Obo mine in Northern China. These were then compared. They normalised the impact on several aspects, and total value was used to measure the environmental impact. The environmental impact of LREEs was 60% lower for the European mine, and 80% lower for HREE. This is mainly because of better emission control, as well as waste and sludge treatment. The reason for this is more stringent regulation in Sweden.

Rare-earth elements in food

REEs can enter our food chains through the soil and water, because plants absorb these REE's. The risk is particularly high for agricultural land close to mines and refineries. REEs have a few benefits, but these are low compared to the high amount of negative effects. REEs damage the metabolic system of the brain, breasts, lungs, kidneys, bones and testes in humans (Adeel et al., 2019).

Future outlook

Chinese researchers found a potential new and significantly more efficient technique to extract the REEs from the minerals. This technique would also be less costly and less polluting. The technique was developed under the lead of scientist Sun Xiaoqi, with substantial backing from the Chinese government. They would use a new material, developed by the scientist's team, which is able to extract the REE in just 20 minutes, whereas it currently takes weeks with the normal techniques. Furthermore, it could be used to extract REE from leached waste and mining debris. The new material used to extract this could also be recycled. The new technology is currently at the industrial testing stage (Chen S. , 2019).

2.2. Rare-earth elements in wind turbines

There are 2 main types of wind turbine generators: doubly-fed induction generators (DFIG) and permanent magnet synchronous generators (PMSG).

The PMSG are based on NdFeB (Neodymium Iron Bohr) magnets. The advantage of this is a lighter and more compact design and greater efficiency at low rotation speeds. The lighter and more compact design leads to the fact that PMSG are used more often for bigger turbines. The fact that they are more efficient at low blade-rotation speeds allows for the elimination of the gearbox, which is normally used in wind turbines to transform the low rotational speed of the blades into a higher rotational speed. When the gearbox is eliminated, this is called direct drive (DD) PMSG (Pavel et al., 2017). The advantage of DD PMSG turbines is that they are more reliable, because of the elimination of the gearbox. Turbulence can put stress on gearboxes, resulting in the gearbox being more likely to fail. This causes geared turbines to require more maintenance. DD PMSG turbines are mostly used in areas which are difficult to access and in areas with high wind speeds. In areas with high wind speeds, the geared turbines are even more likely to fail because even more stress is put on the gearboxes (Arrobas et al., 2017).

Wind turbines using DFIG are based on coil-driven magnets and use a substantial amount of copper. PMSG wind turbines use permanent magnets instead of coil-driven magnets (Arrobas, Hund, Mccormick, Ningthoujam & Drexhage, 2017). Permanent magnets used in PMSG tend to contain a significant amount of REEs, particularly DD PMSG, since the low speed requires a bigger generator. At the same time, there are no, or only a small amount of REEs used in turbines using a DFIG. The REE content could be as high as 246kg per MW for DD PMGS. It is also said that there aren't any real substitutes for NdFeB magnets which are of a similar quality and today, they're considered to be, by far, the best application when using permanent magnets (Chen A. , 2019). This increases reliance on REEs. Other disadvantages of PMSG turbines are higher manufacturing costs and the fact that they require a more expensive converter than geared DFIG turbines (Pavel et al., 2017).

2.2.1. Permanent magnet manufacturers

In 1984, NdFeB magnets were invented by 2 companies simultaneously: The Japanese company Sumitomo and the American company General Motors, both using a slightly different production technique (Lucas et al., 2014).

General motors spun off its magnet production, and after a long history of M&A, joint ventures and a restructuring, the spun off company now operates under the name of Neo Materials. In 2012, Neo Materials was acquired by Mylocorp, the owner of the Mountain Pass mine in the US. However, Mylocorp filed for bankruptcy and, after restructuring, the company operates again under the name: Neo Materials.

Sumitomo became part of Hitachi Corporation. According to the US Department of Energy (2011), the production of the highest quality NdFeB magnets is patented by Hitachi by over 600 patents. These patents comprise both process and component patents. Some of these patents have started to expire in 2014, but certain key patents still continue well past 2014. It is mentioned that there are 8 licensees in China, 2 in Japan and 2 in Germany and 1 in the United Kingdom (Hitachi Metals, 2013). Appendix 22 also provides an overview of the licensees of Hitachi. Hitachi also acquired the wind turbine business of Fuji Heavy Metals in 2012 (NAW Staff, 2012).

In Europe, the manufacturing of REE-magnets for wind turbines is limited to a few firms, with one who is much bigger than all others: Neorem, which is part of Vacuumschmelze (GEUS, 2017). This company is one of the companies licensed by Hitachi Metals to manufacture NdFeB magnets (Neorem, sd). While this is a European manufacturer, it has a subsidiary in China where certain magnets are produced. According to Neorem's website, permanent magnets are produced in Finland, starting from the magnet alloy.

While an American and Japanese company invented and patented NdFeB magnets, it appears like today, production today is dominated by China. Dong et al. (2017) point out that China initially produced lower-grade magnets, but the technology in China improved and they are now able to produce high-quality magnets as well. They also mention that the production of sintered NdFeB magnets increased from 6.500 tonnes in 2001 to 126.300 tonnes in 2015 and there is considered to be an overproduction. Furthermore, they also point out that, based on incomplete statistics, the production in China could be as high as 300.000 ton. This is most likely overly optimistic, given the world-wide rare-earth production is only 170.000 tons and Neodymium makes up over 30% of the weight (US Geological Survey, 2019; US Department of Energy, 2011). According to Benecki (2017), 80% of permanent magnets were produced in China this is expected to continue, given the labor cost advantage and government encouragement. Benecki (2017) identifies 1350 companies active in the permanent magnet production industry, of which 670 are active in China. In the Chinese city Ningbo alone, there are over 200 permanent magnet companies.

There have been multiple issues regarding lawsuits against Hitachi for abuse of their patents or Hitachi suing companies for patent Infringement. For example, in 2012, Hitachi sued 29 firms for infringing their patents and it ended with settlement agreements (Chu, 2016). Another example is 7 Chinese producers of rare-earth magnets who sued Hitachi over 2 of its patents, and in 2016, the patents were found to be partially invalid (Chu, 2016).

For the permanent magnet industry, Benecki (2017) identifies magnet manufacturers, distributors and fabricators. The distributors are quite straightforward: they buy magnets, locally stock them and sell them, sometimes providing engineering assistance. Manufacturers are the companies who produce basic permanent magnet materials. These are then sold to distributors or fabricators. Fabricators buy basic magnets and add value to these magnets.

2.2.2. Demand for permanent magnets in wind turbines

	Weight/ Total weight	Weight (kg)
Neodymium	31,0%	186
Dysprosium	4,1%	25
Praseodymium	5,8%	35
Total REE	**40,9%**	**246**
Other materials	59,1%	354
Total	**100,0%**	**600**

Table 13: Rare-earth content per MW in NdFeB magnets for wind turbines (US Department of Energy, 2012; The World Bank Group, 2017)

In general, permanent magnets consist of 25-35% REEs, 1% Bohr, with the rest being transition metals, most of which is iron (Önal, 2017). According to the US Department of Energy (2011), the permanent magnets in wind turbines weigh around 600kg per MW. They estimate the weight of Neodymium at 31%, Dysprosium 4,1% and praseodymium 0%, since the praseodymium is very low, and it can be substituted. These estimates, except for praseodymium, are very close to the high case estimates of The World Bank. However, The World Bank's high estimate of 35kg per MW for praseodymium is added to table 13 (The World Bank Group, 2017). The total rare-earth content per MW is 246kg (table 13). This is on the high side, but still in line with other sources. For example, Wallington et al. (2013) estimate that wind turbines with NdFeB magnets, contain 171 kg of REE per MW.

At the moment, most wind turbines don't use REEs, given their lower cost of DFIG turbines (Arrobas et al., 2017). Dong et al. (2017) estimate that 25% of wind turbines in China use permanent magnets, whereas this is only 3% in the rest of the world. The US department of energy (2011) mentions similar estimates. However, the share of REEs appears to increase as wind turbines increase in size. In 2016, of the 10 most powerful wind turbine models, 7 use REEs (Pitron, 2018). As wind turbines increase in size and as more and more wind turbines are located offshore, the share of permanent magnets used in wind turbines is likely to increase in the future. For its projections, the US Department of Energy (2011) estimates permanent magnet penetration rates of 15% for onshore turbines and 25% for offshore turbines in the low permanent magnet penetration scenario, and 75% for on- and offshore turbines in the high permanent magnet penetration scenario. Habib & Wenzel (2014) estimate penetration rates of direct drive turbines by 2050 between 25% and 50%.

Through the estimates in table 13, it might appear like Neodymium is the most critical element. However, dysprosium is considered to be the rarest element of the elements used in the permanent magnets, followed by Praseodymium and Neodymium is the least rare. As appendix 19 shows, Neodymium is almost 8 times more common than Dysprosium. Dysprosium is used for its increased resistance to demagnetization at higher temperatures, which is required for permanent magnets in motors or generators (US Department of Energy, 2011). According to Habib & Wenzel (2014), China accounts for 48% of total REE reserves[5], 53% of neodymium reserves and 72% of dysprosium reserves. The fact that such a high percentage of dysprosium reserves are located in China, increased demand for Dysprosium results in increased reliance on China.

A combination of an increase in wind turbine installations and increased use of REEs in wind turbines, could lead to disruptive demand for permanent magnets, given the immense REE content per MW. Van Exter, Bosch, Schipper & Sprecher (2018) studied the impact of the demand for wind turbines and solar panels required for the Netherlands to reach the goals stipulated in the Dutch climate agreement (Klimaatakkoord). They found that, the share of the Netherlands in annual demand of 2030 for Neodymium, Dysprosium and Praseodymium would be respectively 1,8%, 0,75% and 0,9% of the global production in 2017. For the global demand, it is based on scenario's in line with the Paris agreement. They found that the demand in 2030 would exceed 2017 production more than 7 times for Neodymium, and more than 3 times for Dysprosium and Praseodymium (appendix 19). Furthermore, these estimates are only based on demand for wind turbines. In 2010, only 1% of neodymium and dysprosium was used for wind turbines (Habib & Wenzel, 2014). There is still need for these REEs in many other applications, such as hybrid & electric vehicles, consumer electronics, electronic bicycles, magnetic refrigeration etc. (Benecki, 2017). For some of these applications, such as electric vehicles, demand is also expected to increase significantly.

[5] A more recent study estimates China's share in total REE reserves at 37% (US Geological Survey, 2019)

Van Exter et al. (2018) point out that current supply of REEs is not enough. Habib & Wezel (2014) confirm this: in each of their 4 scenario's, business as usual development of REEs doesn't meet demand by 2050. This results in the need for increased production. As already mentioned, there are more than enough reserves of REEs in the world. Both Habib & Wezel (2016) and Van Exter et al. (2018) confirm that the amount of reserves is not particularly the issue. Habib & Wezel (2016) even point out that reserves are more than 100 times the need for Neodymium and 80 times for Dysprosium in their ultimate renewable energy scenario. However, both aforementioned sources identify the bottleneck to be time. The supply might not be able to increase as fast as demand. It takes between 10 and 20 years to open a new mine, resulting in supply not being able to meet sudden increases in demand (Van Exter et al, 2018). As already mentioned, there is a trend to open mines outside China. According to Habib & Wezel (2016), there are approximately 200 ongoing exploration projects outside China. We already mentioned some mines which would already open in 2020, and this could substantially contribute to increased supply.

2.2.3. Reducing reliance on REEs

Another issue is the balance issue (Binnemans & Jones, 2015). Since REEs are almost always co-mined, the supply of each element relative to the other elements remains somewhat constant. This means that, if there is high demand for some elements and mining increases, there could be oversupply of other elements. Given the potential disruptive demand for the REEs used in renewable energy, the balance issue emerges. Some solutions to the balance problem are recycling, substitution and reduced use. These are not only solutions to the balance issue, but also the risk of demand exceeding supply in general.

Substitution and reduced use

In terms of substitution, options are DFIG turbines, superconducting turbines and turbines with permanent magnets other than NdFeB. Magnets. The most straightforward substitute is DFIG wind turbines and they are widely used today. Another substitute could be superconducting wind turbines. The advantage of superconducting wind turbines is that it allows for even bigger wind turbines. When increasing the size of a wind turbine, the generator gets even bigger and heavier and eventually it reaches technical barriers. Superconductors could replace the magnets with lighter electromagnets made from coils of superconducting wire. These superconducting wires would be made out of magnesium diboride (Moore, 2018). In 2018, Ecoswing built the first full scale superconductor wind turbine (Wang, 2018).

There are also some ways to reduce the use of REEs, such as hybrid drive wind turbines and Cerium batteries. Hybrid drive wind turbines use a permanent magnet generator in conjunction with a geared drive. This means that they need less REEs, but also require gearing which increase maintenance cost (US Department of Energy, 2011). According to Pavel et al. (2017), attaching PMSG to a gear that rotates at mid- or high speed, reduces the permanent magnet weight to 160kg and 80kg respectively. For a DD PMSG, this is 600kg, as mentioned earlier. The cerium magnet is one of the main research topics in China related to permanent magnets (Dong et al., 2017). Cerium is 50% more common than Neodymium (appendix 19), which helps in solving the balance problem.

Recycling

Jowitt et al. (2018) report recycling rates of less than 1% for REEs. They also mention that the majority of REE recycling comes from permanent magnets, but that this is still relatively low. The main reasons for low recycling according to them are small amounts in the end-product, lack of economic incentives due to low prices and the design of the products which make recycling difficult. An example of how lack of economic incentives affects this, is Solvay. They started recycling REEs from Fluorescent lamp phosphors in 2012 but discontinued this operation in 2016 because of low REE prices (GEUS, 2017).

Sprecher et al. (2014) compare the environmental impact of recycling of Neodymium in magnets of computer hard disk drives to mining Neodymium from scratch. They found that, once the magnets are hand-picked out of the hard-drives, the recycling process is easy, and the environmental impact is drastically lower than mining new Neodymium. After hand-picking the magnets, they are put into hydrogen gas, causing the magnets to disintegrate into a powder. Subsequently, the powder could be sieved to remove the fragments from the magnet's coating. Eventually, the powder left is equivalent to mined material when it's midway in the manufacturing process of a magnet. The drawback of this method is that hand-picking the magnets from the hard drives is highly labour-intensive. However, for wind-turbines, the magnets are substantially bigger, and this mitigates the issue of recycling being labour-intensive. They also identified shredding the hard-drives as an alternative recycling option to hand-picking the magnets, but this causes a lot of the neodymium to be lost and it's not relevant for wind turbines. EURARE (GEUS, 2017) similarly points out that recycling rates will increase as more big magnets are used because of increased use in wind turbines and hybrid electric vehicles.

Önal (2017) found that a combination of pyro- and hydrometallurgical methods allowed extraction of more than 95% of REEs, with a purity of at least 98%. However, the recovery of iron was less than 1%. Kumari et al. (2018) also found 98% recovery of REEs through roasting.

While recycling is an excellent long-term solution, in the short term it doesn't contribute a lot. When the climate agreement goals want to be met, the market needs to grow rapidly, and the lifetime of wind turbines is long. Given that most wind turbines don't contain REEs, the amount of recyclable permanent magnets from wind turbines is low. In the model of Habib & Wenzel (2014), which also considers demand of other sectors, recycling would be able to meet more than 50% of future demand by 2050. Before 2050, they found that recycling would reduce the supply deficit with 29% for Neodymium and 28% for Dysprosium. For this, they assumed 90% recycling of REE wind turbines, 70% for EV's and 40% in other sectors.

2.2.4. Cost of REEs in wind turbines

	Price/kg	kg/MW	price/MW	cost/MWh
Neodymium	€ 57,06	186	€ 10.613,16	€ 0,35
Dysprosium	€ 288,00	25	€ 7.084,80	€ 0,23
Praseodimium	€ 85,20	35	€ 2.982,00	€ 0,10
Total	€ 84,20	246	€ 20.679,96	€ 0,67

Table 14: Calculation cost of REEs per MWh for onshore wind turbines in 2019

Table 14 shows the calculation of the cost per MWh for the REEs present in the permanent magnets of direct drive onshore wind turbines. The price per kg of REEs is retrieved from SMM (2019). The total cost is 0,67 €/MWh. The LCOE we obtained for onshore wind was 68 €/MWh. This means that the raw material cost of REEs in wind turbines is less than 1%. This is relatively low, and we can assume that the cost of wind turbines isn't very sensitive to changes is REE prices. Given higher capacity factors of offshore wind turbines, more MWh are produced per MW and the cost per MWh will be even lower. The LCOE is also higher for offshore wind turbines, and as a result, the cost of REEs will be well below 1% of the LCOE. A sensitivity analysis to the prices of Neodymium and Dysprosium is included in appendix 24. Moore (2018) reports 25 €/kg as the price for permanent magnets for advanced turbines. Based on this price, the magnet cost is €15.000 €/MW. This signals that the results presented in table 15 might be an overestimation, and the sensitivity to REE prices could be lower in reality.

	Price/kg	kg/MW	price/MW	cost/MWh
Neodymium	€ 304,20	186	€ 56.581,20	€ 1,84
Dysprosium	€ 2.035,80	25	€ 50.080,68	€ 1,63
Praseodimium	€ 207,00	35	€ 7.245,00	€ 0,24
Total	€ 463,79	246	€ 113.906,88	€ 3,71

Table 15:: Calculation cost of REEs per MWh for onshore wind turbines in 2011

Table 15 shows the cost per MWh, using the peak prices of rare-earth crisis in 2011 (Frik, 2012). For dysprosium, the price is over 7 times higher than the current price. In this case, the cost per MWh is €3,71. This is over 5% of the LCOE. This situation clearly shows that disruptive changes in demand or supply can certainly have an impact on the LCOE for wind turbines. However, this was only a short-term effect as prices rapidly dropped afterwards and were back to normal again in 2013 (Sanderson, 2017).

2.2.5. Carbon footprint of REEs in wind turbines

	Neodymium	Dysprosium	Praseodymium
kgCO_2-Eq per kg REE	66,09	738,45	81,53
kg REE per MW	186	25	35
gCO_2-Eq per MW	12.292.740	18.165.870	2.853.550
kWh per MW over lifetime	48.180.000	48.180.000	48.180.000
gCO2-Eq per kWh	0,26	0,38	0,06
Total gCO2-Eq per kWh	0,69		

Table 16: carbon footprint of REEs in an onshore wind turbine

Table 16 shows the carbon footprint of REEs per kWh. It is clear that the carbon footprint of REEs is relatively low. We mentioned earlier that the carbon footprint for onshore wind energy is 9 gCO_2/kWh. For coal, this is over 700 gCO_2/kWh. In comparison, 0,69 gCO_2/kWh is almost negligible. The energy output per MW is higher for offshore wind turbines, which leads to an even lower carbon footprint than our estimate in table 16. As a result, when it comes to global warming, carbon emissions due to rare-earth mining shouldn't be considered to be an impediment to use REEs in wind turbines. However, it shouldn't be forgotten that the carbon footprint is only one of the negative environmental impacts of REEs.

The reason why the share of REES in the total carbon footprint is relatively small because weight REEs is relatively small compared to other materials. While permanent magnets weigh 600kg per MW, it is estimated that a wind turbine contains between 103 and 115 tonnes of steel per MW (The World Bank Group, 2017).

Van Exter et al. (2018) emphasise the importance of REEs to save energy. For example, Niobium enhances the strength of steel, and thus less steel is required for a certain carrying capacity. Because less steel is required, energy is saved on producing steel. It's the same story for lightning. REEs are used in LED-lights, which have a higher energy efficiency than regular light bulbs.

2.3. Critical elements in solar panels

Criteria	Indicators
Risk of supply reduction	Static Reach Reserves Static Reach Resources End-of-Life Recycling Rate
Risk of demand increase	By-Product Dependence Future Technology Demand Substitutability
Concentration risk	Country Concentration Company Concentration
Political risk	Political Stability Policy Perception Regulation Risk

Table 17: Supply risk criteria and indicators considered (Helbig, Bradshaw, Kolotzek, Thorenz, & Tuma, 2016)

A comprehensive study is conducted by Helbig, Bradshaw, Kolotzek, Thorenz, & Tuma (2016), which covers supply risk based on 4 risk criteria and 11 indicators of these criteria (Table 17). In this research, the weighted average risks based on different weighting methods is used at technology level and material level. The technologies considered are Cadmium Telluride (CdTe) and Copper Indium Gallium Selenide (CIGS) solar cells. At material level, following elements will be considered: Cd, Te, Cu, In, Ga, Se and Mo. CdTe and CIGS solar cells are the chosen technologies because they have the biggest share of installed capacity after c-Si panels and the share is expected to increase in the future (Helbig et al., 2016).

In looking at the risk of supply, reserves and resources are considered. Reserves are the amounts of natural stocks of which extraction is technically and economically feasible at the present moment. An example is currently operating mines. Resources are all-natural stocks, of which extraction is potentially feasible in the long run, such as resources which can be mined in the future. Static reach reserves are the ratio of reserves to annual primary production and static reach resources is the ratio of resources to annual primary production. Also, the end-of-recycling-rate is included in risk of supply reduction, because recycling reduces the use of reserves and resources. In appendix 25 and appendix 26, all the indicators, their description, units of measurements, min-max values and standardisation procedures are shown.

Next, there is the normalisation of indexes between 0 and 100 to allow comparison and averaging 11 supply risk indicators. Normalisation methods are also shown in appendix 25 and appendix 26. Lower normalisation values correspond lower risks.

In the next step after normalising and giving scores, different weights were given to each indicator and category on element level by 10 international experts from research and industry according to their importance for overall supply risk. The method is called Analytic Hierarchy Process (AHP). Additionally, different weighting methods are used, such as giving equal weighting to all indicators to compare the results of different weighting methods (Helbig et al., 2016)

As we said before, on element level and technology level (Cd, Te for CdTe and Cu, In, Ga, Se and Mo for CIGS) weighted average total supply risk is calculated. On element level AHP score of each element is calculated. On technology level weighted average of element scores are used. For weighting elements on technology level, equal weighting approach, mass share approach, value approach and maximum approach for weighing are used (Helbig et al., 2016).

2.3.1. Supply risk at element level

The results of all indicators at element level for each element are shown through appendix 27 to appendix 30. For example, static reach reserves range from 23 years for indium to 3182 years for gallium and static reach of resources reach from 73 years for molybdenum to 6250 years for gallium. Normalized values for all indicators are shown on appendix 27. As mentioned above, the relative weighting of the eleven supply risk indicators for the case of thin-film photovoltaics was performed via an Analytic Hierarchy Process (AHP) involving ten international experts. Average of the weights then used to calculate overall risk indicators for each element as given at appendix 31.

Using supply risk indicators for each element, normalization routines and AHP weights, we obtain overall risk values for seven elements considered, which are the bars in figure 24.

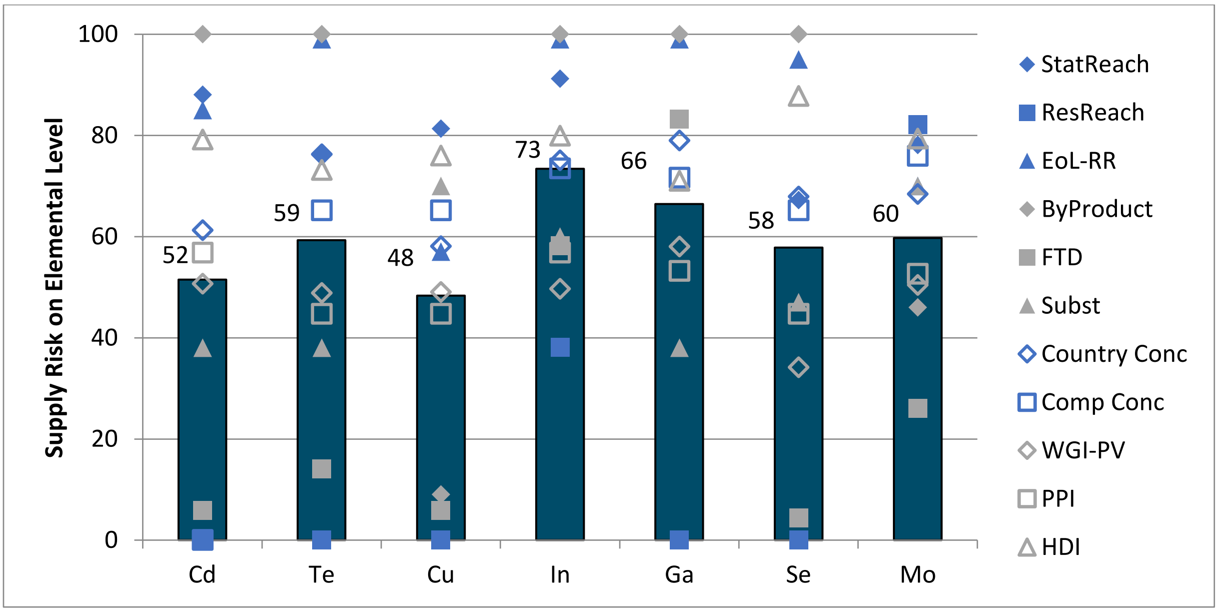

Figure 24: Overall supply risk of elements

Indium shows the highest overall value (73) whereas copper shows lowest (48) one. The highest value of indium results from low static reach, low recycling rates, extraction as a by-product and the highest policy perception. Copper on the other hand has high static reach, the highest recycling rate and the lowest product dependence. Comparison with other weighting methods is shown in appendix 34. It demonstrates that higher supply risk is obtained with AHP weighting except molybdenum. However, overall order of supply risk scores remains the same.

2.3.2. Supply risk at technology level

Overall supply risk on the technology level is calculated by weighing respective materials in these technologies using arithmetic means, mass share, cost share[6] (appendix 35, 36) and maximum (considers only highest supply risk element in technology) which is determined by Te in CdTe and In for CIGS (Figure 25).

[6] This is calculated as the mass multiplied by the price, and this is divided by the total mass multiplied by the price

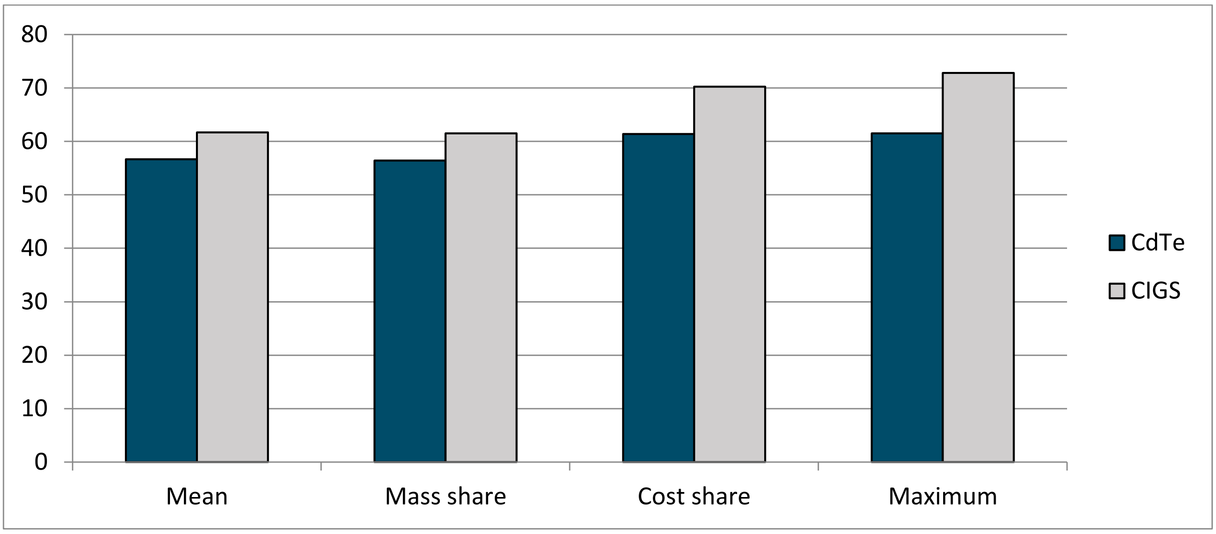

Figure 25: Overall supply risk by technologies using different weighting methods

In any case it is clear that CdTe is the less risky technology than CIGS. This result should not be seen as a physical expression of scarcity but rather a relative expression of mid-to-long term supply risks (Helbig et al., 2016).

2.3.3. Comparison supply and demand

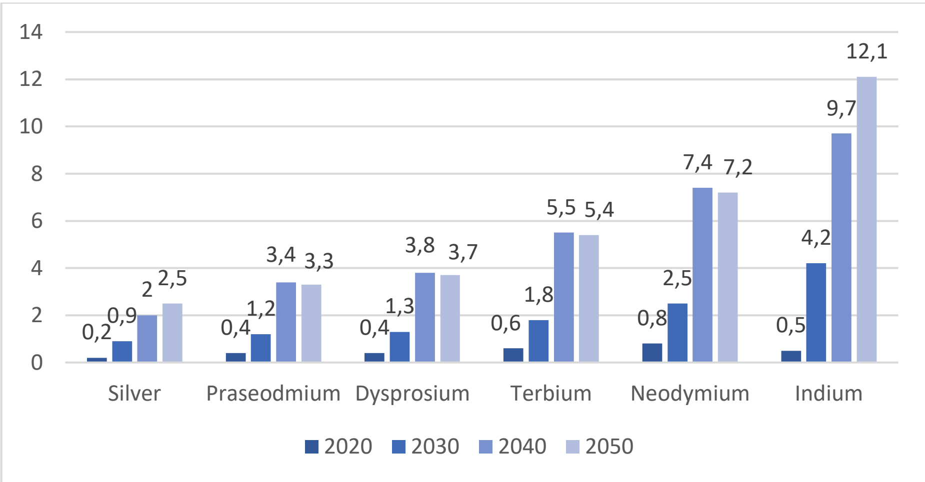

Figure 26: Annual global critical metal demand for wind and PV panels between 2020 and 2050, compared to the annual metal production index (2017=1)

Van Exter et al. (2018) also conducted analyses for other metals than REEs. Based on the Paris agreement on climate change, they calculated metal demand used in renewable energy technologies. They concluded that if rest of the world would follow similar green energy targets with the Netherlands by 2050, production of some most critical metals would need to increase several fold by 2050, including more than 12 times for Indium (Figure 26). With the current level of production and recycling rates, it might not be possible to fully implement the green energy transition in line with Paris accord. To transition to a circular economy, that is all the material requirements for renewable energy can be provided through recycling, recycling rates should be very high. However, in the short term, as solar and wind capacity increases exponentially, even with very high recycling rates demand will still be dependent on mining because most of the capacity is installed recent years and it takes on average 25 years for solar panels and wind turbines to retire.

2.4. Critical elements in energy storage

Given the future expected increase in battery production, raw materials and supply chain security has a great importance for the European Union. Essential materials for battery production are identified as cobalt, lithium, manganese and graphite. Sourcing of the four essential battery raw materials is very concentrated in only a few countries. For example, the Democratic Republic of Congo supplies 64% of world's cobalt and China produces 69% of graphite worldwide. Similarly, 36% of lithium and 20% of manganese comes from South Africa. Europe mainly imports raw materials for batteries with the exception of Finland which provides 66% of EU demand (European Comission, 2018).

It is expected that by 2025, cobalt demand, only for rechargeable batteries, will be 3 times higher than 2018, surpassing the total production of 2018. For lithium, global demand for rechargeable batteries will be 3,5 times higher than current levels and 75% of world lithium production will be used in rechargeable batteries (European Comission, 2018).

According to latest outlooks, the cobalt supply chain is at risk due to the high concentration in the Democratic republic of Congo, as well as the fact that refining cobalt predominantly takes place in China. Rapid increases in demand and inflexibility of supply, as well as political risk might disrupt the cobalt supply chain in the future (European Comission, 2018).

Li-ion batteries consumed around 40% of the global lithium carbonate equivalent (LCE) in 2015, of which 14% is used in electric batteries. Projections for 2025 indicate that electric vehicles alone will use 200.000 tonnes of LCE, which equates total current global LCE supply. Even without recycling, known lithium reserves are enough to meet the demand increase but there are only a few lithium processors with the ability to process high grade lithium compounds that batteries need (European Comission, 2018).

IV. Conclusion

This report started with an overview of the installed capacity over time per region. Particularly wind energy and solar energy capacity increased drastically during the last decade. Through the first chapter, it became clear that low costs in combination with low carbon emissions allowed it this rapid growth. In the second chapter, it was found that rare-earth elements and other critical element's shouldn't necessarily hinder this growth in the future.

In terms of total lifecycle cost of renewable energy, we can conclude that utility scale solar PV has the lowest LCOE, but this is closely followed by the LCOE of onshore wind. We can also conclude that offshore wind has the highest LCOE, but the gap with onshore wind can close in the future due to technology improvements, for example, through floating wind turbines and superconducting wind turbines. Compared to combined cycle gas turbines, the LCOE of onshore wind and utility-scale PV is lower when taking the social cost of carbon into account. When we don't take this into account, combined-cycle gas turbines are still cheaper when used at optimal capacity. However, since they are not used at optimal capacity in practice, onshore wind and solar energy are cheaper.

The increased cost of combined cycle gas turbines when not used at optimal capacity is linked to the balance of system costs. These increase when the penetration rate of wind energy and solar energy increases. When using wind energy and solar energy together, the variability in energy supply is able to offset each other to some extent, but the variability will still be significant. Because of this, other energy facilities or storage facilities are still required for times when demand exceeds supply. As a result, while wind and solar energy have the lowest LCOE when including the social cost of carbon, non-renewable energy sources or batteries are still needed for balancing specifically when implemented in large-scale. For batteries, the cost appears to be highly dependent on the study conducted. We can conclude that the optimal mix includes wind and solar energy, as well as batteries or other energy sources as back-up facilities. However, when the cost decreases further for renewable energy and energy storage, it might be possible to fully rely on renewable energy in the future.

For the end-of-life costs, it appears like they are highly dependent on the technology used. We can conclude that they are negligible for onshore wind, substantial for offshore wind and born by the manufactures for solar panels. In terms of recycling practices, the most important materials in wind turbines are recycled, except for the blades. However, technological improvements might make recycling of turbine blades more relevant in the future. For solar energy, although most of the materials of PV panels such as aluminium, glass and polymers are recycled, The recycling rate of some critical metals is very low but expected to increase in the future, as new dedicated recycling plants come online and panels reaching end-of-life stage grows exponentially, therefore making these recycling investments economically viable

We also can conclude that solar panels don't contain rare-earth elements and some wind turbines models do. There is potential for price increases through reductions in supply by China or increases in demand of REEs for wind turbines. However, it is also clear that the LCOE of wind turbines isn't very sensitive to REE prices. Nonetheless, when prices increase as much as during the rare-earth crisis is 2011, the impact would be significant.

In the case of price increases, there are enough substitutes. Furthermore, these price peaks would only be temporary, since reserves are plentiful and opening new mines would drive prices down again in the long term. There also is a clear trend in which the production in China relative to other countries decreases. This trend is expected to continue, with several new mines outside China which are currently being explored. This could further reduce risk of price increases.

Opening mines outside China could also positively affect the environmental impact of rare-earth elements, as research signals that the environmental impact of a European mine could be 80% lower for certain REEs compared to China. REE mining and processing has significant negative environmental aspects, mainly related to human toxicity, aquatic ecotoxicity, eutrophication of fresh water and particulate matter. The impact of REEs on the carbon footprint of wind turbines is relatively small and rare-earth elements. Most likely, the positive impact is bigger than the negative regarding climate change, since they can also improve energy-efficiency.

For the critical elements used PV panels using different supply chain risk categories, we conclude that some elements are at risk due several possible supply chain disruptions, such as an increase in future technology demand, concentration risk, political risk, by-product dependence of mining these elements etc. It is critical to increase mining efforts in the short term in other parts of the world beyond china, reduce materials used in these technologies and increase recycling rates in order to transform to circular economy and ultimately eliminate our critical element needs from mining. However, if material demand is too high and cannot be supplied for thin-film technologies in the future, crystalline panels and other emerging technologies can be used as an alternative. There are also substitution efforts being done for these elements at R&D level and viable alternatives can be found in the future.

Regarding critical elements used in batteries, there is concentration risk and political risk for the most critical elements used in battery technologies, specifically in the case of lithium ion batteries. These are cobalt, lithium, manganese and graphite. When it comes to cost and economic viability of batteries, we found that it is highly dependent on use cases, the geographies where they are used and revenue streams. Government subsidies and incentive programs also play a big role in economic viability of battery technologies. However, capital cost of batteries is expected to decline further, in line with past trends. This is particularly the case for li-ion technologies due to reduced material usage and widescale adaptation of these technologies, thus paving the way truly renewable and circular economy.

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Appendix

Appendix 1: CAGR renewable energy technologies

Technology	2000	2018	CAGR
Hydropower	783 004	1.292.595	3%
Wind	16 926	563.726	20%
Solar	1 227	485.826	37%
Bioenergy	29 763	115.731	7%
Geothermal	8 236	13.329	3%
Marine	238	532	4%

Source: IRENA (2017)

Appendix 2: Onshore wind energy installed cost calculation

Source	Year	Location	low	high	average	weight
IEA (Hand, 2018)	2016	Germany	1.520	1.520	1.520	0,125
IEA (Hand, 2018)	2016	Europe	1.564	1.564	1.564	0,125
IRENA (2018)	2016	Europe	1.068	2.225	1.647	0,200
ADEME (2017)	2017	France	1.300	1.700	1.500	0,200
DOE (Wiser & Bollinger, 2018)	2017	USA	1.433	1.433	1.433	0,050
NREL (2018)	2018	USA	1.400	1.525	1.462	0,050
ISE (2018)	2018	Germany	1.500	2.000	1.750	0,200
Lazard (2018)	2018	USA	1.024	1.380	1.202	0,050
				Weighted average		1.570

Appendix 3: LCOE onshore wind energy sensitivity analysis

		Capacity factor										
	68	20%	21%	22%	23%	24%	25%	26%	27%	28%	29%	30%
Discount rate	3%	77	73	70	67	64	61	59	57	55	53	51
	4%	82	78	75	71	68	66	63	61	59	57	55
	5%	88	84	80	76	73	70	68	65	63	61	59
	6%	94	89	85	81	78	75	72	69	67	65	62
	7%	100	95	91	87	83	80	77	74	71	69	66
	8%	106	101	96	92	88	85	81	78	76	73	71
	9%	112	107	102	98	93	90	86	83	80	77	75
	10%	119	113	108	103	99	95	91	88	85	82	79

Installed cost ('000 €/MW)	O&M ('000 €/MW per year)										
68	20	25	30	35	40	45	50	55	60	65	70
1.200	48	51	53	55	58	60	62	64	67	69	71
1.300	52	54	56	58	61	63	65	68	70	72	74
1.400	55	57	59	62	64	66	69	71	73	75	78
1.500	58	60	63	65	67	70	72	74	76	79	81
1.570	60	63	65	67	70	72	74	76	79	81	83
1.600	61	64	66	68	71	73	75	77	80	82	84
1.700	65	67	69	72	74	76	78	81	83	85	87
1.800	68	70	72	75	77	79	82	84	86	88	91
1.900	71	73	76	78	80	83	85	87	89	92	94
2.000	74	77	79	81	84	86	88	90	93	95	97

Appendix 4: Carbon emissions per energy technology

Electricity Supply technologies	**Direct emissions g CO_2/kWh min/median/max**	**Lifecycle emissions g CO_2/kWh min/median/max**
Coal - PC	670/**760**/870	740/**820**/910
Gas - Combined cycle	350/**370**/490	410/**490**/650
Biomass - Cofiring	n/a	620/**740**/890
Biomass- Dedicated	n/a	130/**230**/420
Geothermal	0	6/**38**/79
Hydropower	0	1/**24**/220
Nuclear	0	3.7/**12**/110
Concentrated solar	0	8.8/**27**/63
Solar power - rooftop	0	26/**41**/60
Solar power - utility	0	18/**48**/180
Onshore wind	0	7.0/**11**/56
Offshore wind	0	8.0/**12**/35

Source: Schlömer et al. (2014)

Appendix 5: Offshore wind energy installed cost breakdown

	'000€/ MW	% of Total
Turbine	1.300	38%
Turbine installation and commissioning	162	5%
Turbine subtotal	**1.462**	**42%**
Foundations supply	551	16%
Foundations installation	210	6%
Foundations Subtotal	**762**	**22%**
Array cable supply	43	1%
Array cable installation	117	3%
Offshore substation	185	5%
Export cable supply	138	4%
Export cable installation	62	2%
Land-based substation and grid connection	93	3%
Electrical infrastructure Subtotal	**638**	**18%**
Construction insurance	45	1%
Project management	145	4%
Contingency	305	9%
Other CAPEX Subtotal	**496**	**14%**
Total Construction CAPEX	**3.357**	**97%**
Development	101	3%
Grand Total	**3.459**	**100%**

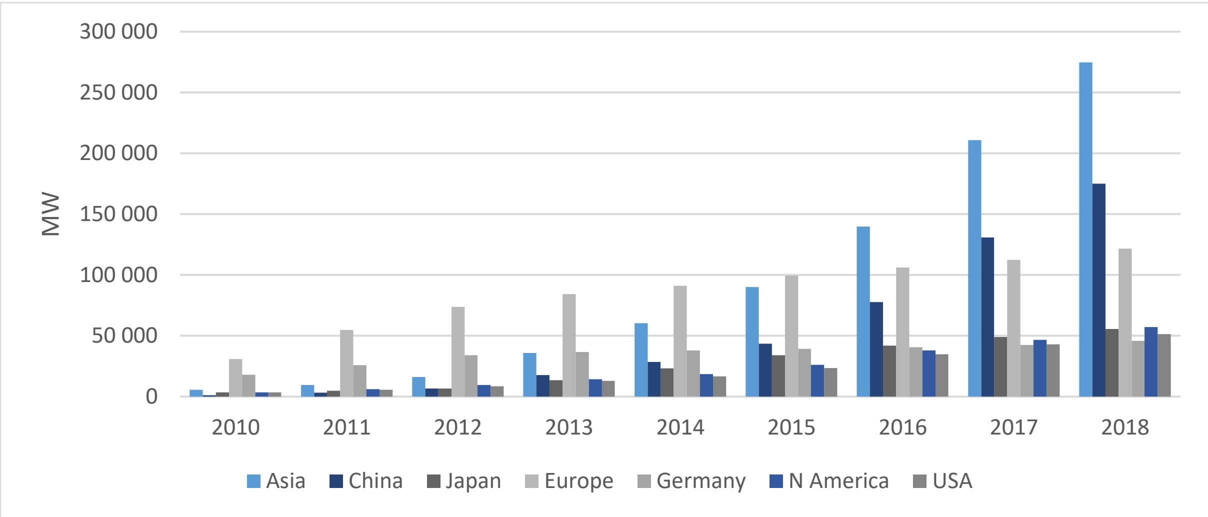
Source: Hand (2018)

Appendix 6: LCOE offshore wind energy sensitivity analysis

		Capacity factor										
	88	32%	34%	36%	38%	40%	42%	44%	46%	48%	50%	52%
Discount rate	3%	104	98	93	88	84	80	76	73	70	67	64
	4%	111	105	99	94	89	85	81	77	74	71	69
	5%	119	112	105	100	95	90	86	83	79	76	73
	6%	126	119	112	106	101	96	92	88	84	81	78
	7%	134	126	119	113	107	102	98	93	89	86	83
	8%	142	134	127	120	114	108	104	99	95	91	88
	9%	151	142	134	127	121	115	110	105	101	97	93
	10%	160	150	142	134	128	122	116	111	106	102	98

		O&M ('000 €/MW per year)										
	88	45	50	55	60	65	70	75	80	85	90	95
Installed cost ('000 €/MW)	3.000	73	75	76	77	79	80	81	83	84	86	87
	3.100	75	77	78	79	81	82	83	85	86	87	89
	3.200	77	79	80	81	83	84	85	87	88	89	91
	3.300	79	80	82	83	85	86	87	89	90	91	93
	3.400	81	82	84	85	87	88	89	91	92	93	95
	3.500	83	84	86	87	88	90	91	93	94	95	97
	3.600	85	86	88	89	90	92	93	94	96	97	99
	3.700	87	88	90	91	92	94	95	96	98	99	100
	3.800	89	90	92	93	94	96	97	98	100	101	102
	3.900	91	92	93	95	96	98	99	100	102	103	104

Appendix 7: Evolution of cumulative installed PV capacity by region over time



Source: IRENA (2019)

Appendix 8: Calculation LCOE for utility scale PV

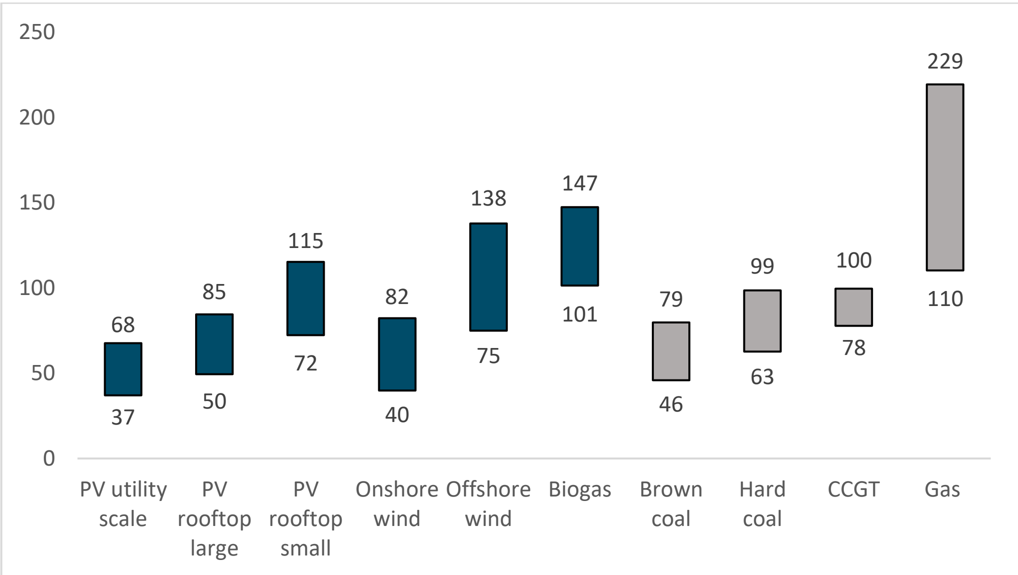
Years	0	1	2	3	4	5	...	25
Installed cost	700	-	-	-	-	-	...	0
O&M		18	18	18	18	18	...	18
Total cost	700	18	18	18	18	18	...	18
Discounted Cost	700	17	16	16	15	14	...	6
Generated energy(kWh)		1.102	1.099	1.097	1.094	1.091	...	1.038
Discounted energy (kWh)		1.078	1.028	981	936	893	...	348
Total Discounted Cost	964							
Net present energy	16.295							
LCOE (€/kWh)	0,06							
LCOE (€/MWh)	59,28							

Appendix 9: LCOE utility scale PV sensitivity analysis

Discount rate	Irradiance level										
59	700	800	900	1000	1100	1200	1300	1400	1500	1600	1700
3%	84	73	65	59	53	49	45	42	39	37	34
4%	90	79	70	63	57	52	48	45	42	39	37
5%	97	84	75	68	61	56	52	48	45	42	40
6%	103	90	80	72	66	60	56	52	48	45	43
7%	110	97	86	77	70	64	59	55	51	48	45
8%	118	103	91	82	75	69	63	59	55	51	48
9%	125	109	97	87	79	73	67	62	58	55	51
10%	132	116	103	93	84	77	71	66	62	58	55

Installed cost ('000 €/MW)	Operating expenses ('000 €/MW per year)										
59	12	14	16	18	20	22	24	26	28	30	32
400	36	38	39	41	43	45	47	49	51	52	54
500	42	44	46	47	49	51	53	55	57	59	60
600	48	50	52	54	55	57	59	61	63	65	67
700	54	56	58	60	62	63	65	67	69	71	73
800	60	62	64	66	68	70	71	73	75	77	79
900	66	68	70	72	74	76	78	79	81	83	85
1000	73	74	76	78	80	82	84	86	87	89	91
1100	79	81	82	84	86	88	90	92	94	95	97
1200	85	87	89	90	92	94	96	98	100	102	103
1300	91	93	95	97	98	100	102	104	106	108	110

Appendix 10: LCOE of different energy technologies according to Fraunhofer ISE



Source: Fraunhofer ISE (2018)

Appendix 11: Min, max and average values of renewable energy technologies (2010-2017) according to IRENA

2010				2017		
Technology	**min**	**max**	**average**	**min**	**max**	**average**
CSP	260	350	330	160	260	220
Geothermal energy	40	80	50	30	140	70
Hydropower	20	320	40	20	220	50
Offshore wind	100	260	170	110	240	140
Onshore wind	50	210	80	40	280	60
Solar photovoltaic	60	400	360	50	350	100
Solid biomass	40	210	70	50	140	70

Source: IRENA (2017)

Appendix 12: LCOE evolution of energy technologies in US (USD/MWh)

Year	Gas peaker	Nuclear	Coal	Gas - Combined Cycle	Solar PV - Cyrstalline	Wind
2009	275	123	111	83	359	135
2010	243	96	111	82	248	124
2011	227	95	111	83	157	71
2012	216	96	102	75	125	72
2013	205	104	105	74	104	70
2014	205	112	112	74	79	59
2015	192	117	108	64	64	55
2016	191	117	102	63	55	47
2017	183	148	102	60	50	45
2018	179	151	102	58	43	42

Source: Lazard (2018)

Appendix 13: Materials used in different PV technologies

Material used in PV technologies		Aluminium	Glass	Sealants	Copper	Silicon	Compound Semiconductor	Polymer	Silver	Other Metals
c-Si	2014	8%	76%		1%	5%		10%	0.05%	0.05%
	2030	7%	80%			3%		10%	0%	0%
CIGS	2014	7%	89%					4%		trace amounts
	2030	8%	88%					4%		
CdTe	2014		97%				0%	3%		0.26%
	2030		96%				0%	4%		0.41%

Source: IRENA (2016)

Appendix 14: Use Cases and technologies assessed

	Use Cases	Description	Technologies assessed
In-front-of-the-meter	Wholesale	Designed to replace gas turbine facilities to meet rapidly increasing demand at peak	Lithium-ion Flow Battery-Vanadium Flow Battery-Zinc Bromide
	Transmission and Distribution	Energy storage system designed to defer transmission and/or distribution upgrades, typically placed at substations or distribution feeder controlled by utilities to provide flexible capacity while also maintaining grid stability	Lithium-ion Flow Battery-Vanadium Flow Battery-Zinc Bromide
	Utility-Scale (PV+Storage)	Energy storage system designed to be paired with large solar PV facilities to improve the market price of solar generation, reduce solar curtailment and provide grid support when not supporting solar objectives	Lithium-ion Flow Battery-Vanadium Flow Battery-Zinc Bromide
Behind-the-meter	Commercial & Industrial (Standalone)	Energy storage system designed for behind-the-meter peak shaving and demand charge reduction services for commercial energy users	Lithium-ion Lead Acid Advanced Lead (Lead Carbon)
	Commercial & Industrial (PV storage)	Energy storage system designed for behind-the-meter peak shaving and demand charge reduction services for commercial energy users	Lithium-ion Lead Acid Advanced Lead (Lead Carbon)
	Residential (PV + Storage)	Energy storage system designed for behind-the-meter residential home use—provides backup power, power quality improvements and extends usefulness of self-generation (e.g., “solar PV + storage”)	Lithium-ion Lead Acid Advanced Lead (Lead Carbon)

Source: Lazard (2018)

Appendix 15: LCOS of different use cases and battery technologies

	Unsubsidized Levelized Cost of Storage Comparison $/MWh		min (USD/MWh)	max(USD/MWh)
In-front-of-the-meter	Wholesale	Lithium-ion	204	298
		Flow Battery-Vandanium	257	390
		Flow Battery-Zinc Bromide	267	300
	Transmission and Distribution	Lithium-ion	263	471
		Flow Battery-Vandanium	293	467
		Flow Battery-Zinc Bromide	406	464
	Utility-Scale (PV+Storage)	Lithium-ion	108	140
		Flow Battery-Vandanium	133	222
		Flow Battery-Zinc Bromide	115	167
Behind-the-meter	Commercial & Industrial (Standalone)	Lithium-ion	829	1152
		Lead Acid	1076	1225
		Advanced Lead (Lead Carbon)	1005	1204
	Commercial & Industrial (PV storage)	Lithium-ion	315	366
		Lead Acid	382	399
		Advanced Lead (Lead Carbon)	347	378
	Residential (PV + Storage)	Lithium-ion	476	735
		Lead Acid	512	707
		Advanced Lead (Lead Carbon)	498	675

Source: Lazard (2018)

Appendix 16: Revenue streams and descriptions

Revenue streams	Description
Demand Response	Manages high wholesale price or emergency conditions on the grid by calling on users to reduce or shift electricity demand
Energy Arbitrage	Allows storage of inexpensive electricity to sell at a higher price later (includes only wholesale electricity purchase)
Frequency Regulation	Provides immediate (4-second) power to maintain generation load balance and prevent frequency fluctuations
Resource Adequacy	Provides capacity to meet generation requirements at peak loading in a region with limited generation and/or transmission capacity
Spinning/ Non-Spinning Reserves	Maintains electricity output during unexpected contingency event (e.g., an outage) immediately (spinning reserve) or within a short period (non-spinning reserve)
Distribution Deferral	Provide extra capacity to meet projected load growth for the purpose of delaying, reducing or avoiding distribution system investment in a region
Transmission Deferral	Provide extra capacity to meet projected load growth for the purpose of delaying, reducing or avoiding transmission system investment
Bill Management	Allows reduction of demand charge using battery discharge and the daily storage of electricity for use when time of use rates are highest

Source: Lazard (2018)

Appendix 17: IRR and share of IRR for different use cases and locations (US)

	Wholesale	Transmission and Distribution	Utility-Scale (PV+Storage)	C & I (Standalone)	C & I (PV storage)	Residential (PV + Storage)
	California	New York	West Texas	San Fransisco	San Fransisco	Los Angeles
Energy Arbitrage	20%	2%	66%	4%	2%	--
Frequency Regulation	29%	14%	9%	--	--	--
Spinning/Non-Spinning Reserves	8%	13%	26%	12%	5%	--
Resource Adequacy	43%	15%	--	18%	13%	--
Distribution Deferral	--	56%	--	--	--	--
Demand Response–Wholesale	--	--	--	--	--	--
Demand Response–Utility	--	--	--	3%	2%	--
Bill Management	--	--	--	64%	78%	87%
Local Incentive Payments	--	--	--	--	--	13%
Total IRR	16.70%	22.80%	8.80%	11.90%	13.60%	5.20%

Source: Lazard (2018)

Appendix 18: IRR and share of IRR for different use cases and locations (International)

	Wholesale	Transmission and Distribution	Utility-Scale (PV+Storage)	C & I (Standalone)	C & I (PV storage)	Residential (PV + Storage)
	UK	--	Australia	Ontario	Australia	Germany
Energy Arbitrage	--	--	74%	--	--	--
Frequency Regulation	71%	--	5%	--	--	--
Spinning/Non-Spinning Reserves	17%	--	--	--	--	--
Resource Adequacy	12%	--	21%	--	--	--
Distribution Deferral	--	--	--	--	--	--
Demand Response–Wholesale	--	--	--	12%	--	--
Demand Response–Utility	--	--	--	--	--	--
Bill Management	--	--	--	89%	100%	85%
Local Incentive Payments	--	--	--	--	--	15%
Total IRR	4.40%	--	8.70%	20.10%	14.30%	2.50%

Source: Lazard (2018)

Appendix 19: Overview rare-earth elements, including crustal abundance

Light rare-earth elements			
Element	Symbol	Atomic number	Crustal abundance
Lanthanum	La	57	39
Cerium	Ce	58	66,5
Praseodymium	Pr	59	9,2
Neodymium	Nd	60	41,5
Promethium	Pm	61	n/a
Samarium	Sm	62	7,05
Europium	Eu	63	2
Gadolinium	Gd	64	6,2
Heavy rare-earth elements			
Element	Symbol	Atomic number	Crustal abundance
Terbium	Tb	65	1,2
Dysprosium	Dy	66	5,2
Holmium	Ho	67	1,3
Erbium	Er	68	3,5
Thulium	Tm	69	0,52
Ytterbium	Yb	70	3,2
Lutetium	Lu	71	0,8
Scandium	Sc	21	n/a
Yttrium	Yb	39	33

Source: Van Gosen et al. (2014)

Note: Gold, silver, lead and copper have 0,004; 0,075; 14 and 60 parts per million respectively

Appendix 20: Overview rare-earth element production in 2017 and 2018 and reserves per country

country	production 2017	production 2018	Reserves
Australia	19.000	20.000	3.400.000
Brazil	1.700	1.000	22.000.000
Burma (Myanmar)	n/a	5.000	n/a
Burundi	0	1.000	n/a
China	105.000	120.000	44.000.000
India	1.800	1.800	6.900.000
Malaysia	180	200	30.000
Russia	2.600	2.600	120.000.000
Thailand	1.300	1.000	n/a
United States	0	15.000	1.400.000
Vietnam	200	400	22.000.000
Other countries	/	/	4.400.000
World total (rounded)	**132.000**	**170.000**	**120.000.000**

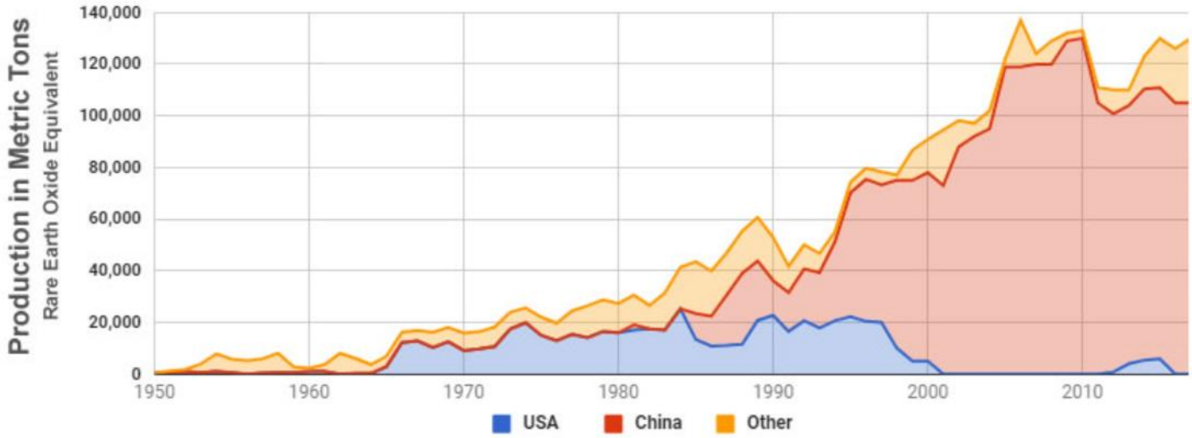
Source: US Geological Survey (2019)

Appendix 21: Overview rare-earth element production in 2017 and 2018 and reserves per country (as a % of world total)

country	production 2017	production 2018	Reserves
Australia	14%	12%	3%
Brazil	1%	1%	18%
Burma (Myanmar)	n/a	3%	n/a
Burundi	0%	1%	n/a
China	80%	71%	37%
India	1%	1%	6%
Malaysia	0%	0%	0%
Russia	2%	2%	10%
Thailand	1%	1%	n/a
United States	0%	9%	1%
Vietnam	0%	0%	18%
Other countries	/	/	4%
World total (rounded)	100%	100%	100%

Source: US Geological Survey (2019)

Appendix 22: Overview historical evolution rare-earth element production



Source: King (sd)

Appendix 23: Hitachi licensees per country

Company name	Country
Neorem Magnets Oy	Germany
Vacuumschmelze GmbH & Co. KG	Germany
Magnetfabrik Schramberg GmbH	Germany
The Morgan Crucible Company plc	United Kingdom
Advanced Technology & Materials Co., Ltd.	China
Anhui Earth-Panda Advance Magnetic Material Co., Ltd	China
Beijing Jingci Magnet Co.	China
Beijing Zhong Ke San Huan High-Tech Co., Ltd.	China
Ningbo Jinji Strong Magnetic Material Co., Ltd.	China
Ningbo Yunsheng Co., Ltd.	China
Thinova Magnet Co., Ltd.	China
Yantai Zhenghai Magnetic Material Co., Ltd.	China
Shin-Etsu Chemical Co., Ltd.	Japan
TDK Corporation	Japan

Source: Hitachi Metals, 2013

Appendix 24: LCOE sensitivity analysis to rare-earth prices for onshore wind energy

Dysprosium price per kg (€) \ Neodymium price per KG (€)	0,67	288	300	400	500	600	700	800	900	1000
	57	0,67	0,68	0,76	0,84	0,92	1,00	1,08	1,16	1,24
	75	0,78	0,79	0,87	0,95	1,03	1,11	1,19	1,27	1,35
	100	0,93	0,94	1,02	1,10	1,18	1,26	1,34	1,42	1,50
	125	1,09	1,09	1,17	1,25	1,34	1,42	1,50	1,58	1,66
	150	1,24	1,25	1,33	1,41	1,49	1,57	1,65	1,73	1,81
	175	1,39	1,40	1,48	1,56	1,64	1,72	1,80	1,88	1,96
	200	1,54	1,55	1,63	1,71	1,79	1,87	1,95	2,03	2,11
	225	1,69	1,70	1,78	1,86	1,94	2,02	2,10	2,18	2,26
	250	1,84	1,85	1,93	2,01	2,09	2,17	2,25	2,33	2,41

Note: this is only for the share of rare-earth elements in the total LCOE. The total LCOE for onshore wind is 68.

Appendix 25: Explanation of supply risk indicators in the categories "risk of supply reduction" and "risk of demand increase"

	Indicator	Indicator Description	Unit	Min	Max	Calculation
Risk of Supply Reduction	Static Reach Reserves	The static reach of the reserves (SR) of a raw material is measured by the ratio of annual primary production to the estimated global reserves. Reserves are the ores currently technically and economically extractable from known deposits. The value gives an indication of the market pressure for further exploration and for the development of new extraction capabilities, possibly leading to higher price levels.	years	0 years	infinite (∞ years)	$S_1 = 100 - 0{,}2\mathrm{SR} - 0{,}008\mathrm{SR}^2$
	Static Reach Resources	The static reach of the resources (RR) of a raw material is measured by the ratio of annual production to estimated global resources. Resources are deposits from which the economic extraction of the ore is potentially feasible, but not at present. The value is the best available measure of a possible physical scarcity of a raw material due to potential mineral depletion, independent of current price levels.	years	0 years	infinite (∞ years)	$S_2 = 100 - 0{,}1\mathrm{RR} - 0{,}002\mathrm{RR}^2$
	EoL-Recycling Rate	The end-of-life recycling rate (EoL-RR) of a raw material is measured by the ratio of current annual recycled material flow to the annual discard rate of the raw material. The value gives an estimate of the amount of available secondary material, which is independent of mining and primary refining activities and can thus smooth out supply disruptions or price peaks.	%	0%	100%	$S_3 = 100 - \mathrm{EoL\text{-}RR}$
Risk of Demand Increase	By-Product Dependence	The by-product dependence (BPD) is the percentage of the element mined as a byproduct of the global production of another element. This is normally the case, when mining solely for the raw material itself is not economically feasible. This figure is a measure of the potential inability to increase primary production in response to an increase in demand.	%	0%	100%	$S_4 = \mathrm{BPD}$
	Future Technology Demand	Future technology demand (FTD) is given by the ratio of expected additional demand in a future year due to new, future technologies and global production in a past year. The value gives an indication of the market pressure for increasing global extraction due to future technologies and therefore of potential additional competition in the commodity markets.	%	0%	infinite (∞ %)	$S_5 = \left(\left(\sqrt[t]{(1+\mathrm{FTD})}\right) - 1\right) * 1000$ t=24 years
	Substitutability	Substitutability (Subst) is an estimate of the extent to which a raw material can be replaced by another raw material, without there being a too great a loss of essential properties. It is gauged by expert assessment. The value gives an estimates of the extent to which demand can be shifted to other materials in case of supply shortage and thus of the potential to smooth out supply disruptions or price peaks.	dimension-less	0	100	$S_6 = 100 - \mathrm{Subst}$

Source: Helbig et al. (2016)

Appendix 26: Explanation of supply risk indicators in the categories “concentration risk” and “political risk”

#	Indicator	Indicator Description	Unit	Min	Max	Calculation
Concentration Risk	Country Concentration	The concentration of the annual production of a raw material at the country level is measured by the Herfindahl-Hirschman Index, which is the sum over the squares of the production shares of the countries in percent. The value indicates directly market concentration in a few countries and thus the possibility of strategic exploitation of a monopolistic position at times of international crisis or dispute.	Herfindahl-Hirschman-Index	0	10000	$S_7 = 21.64\ln(\mathrm{HHI}) - 99.31$
	Company Concentration	The concentration of the annual production of a raw material at the company level is measured by the Herfindahl-Hirschman Index, which is the sum over the squares of the production shares of the companies in percent. The value indicates directly market concentration in a few companies and thus the likelihood of oligopolistic structures, which are linked in turn to higher price levels, low levels of competition and strategic misuse.	Herfindahl-Hirschman-Index	0	10000	$S_8 = 15.81\ln(\mathrm{HHI}) - 45.62$
Political Risk	Country Risk Political Stability	The risk of political instability in producing countries is measured by the Worldwide Governance Indicator for Political Stability and Absence of Violence/Terrorism, presented by the World Bank, weighted by the production share in each producing country. The value is an indication of the likelihood of disruption in production and export in the countries concerned due to unrest, coups d’état, terrorism or other situations involving violence.	Worldwide Governance Indicator – Political Stability and Absence of Violence/Terrorism	-2,5	2,5	$S_9 = 20 * (2.5 - \mathrm{WGI})$
	Country Risk Policy Perception	The indicator Policy Perception is an assessment of the ability of producing countries to implement new mining projects, weighted by the production share in each country. The Policy Perception is evaluated by mining industry experts and summarized by the Fraser Institute. The value is a measure of the ability of the market to continue to function and/or of primary production to increase further based on the rule of law and governance procedures in producing countries.	Policy Perception Index	0	100	$S_{10} = 100 - \mathrm{PPI}$
	Country Risk Regulation	The “regulation risk” attempts to measure the likelihood of the producing countries to actually implement restrictions on mining, refining and trade, as indicated by their level of societal development. This in turn is measured by the HDI (Human Development Index), as presented by the United Nations Development Programme and weighted by the production share in each producing country. The value assesses the likelihood that further mineral extraction and refining activities are prevented due to regulations, taxes, tariffs or taxes in producing countries.	Human Development Index	0	1	$S_{11} = 100 * \mathrm{HDI}$

Appendix 27: Supply risk data on elemental level

Indicator	Dimension	Risk	Cd	Te	Cu	In	Ga	Se	Mo
Static Reach Reserves	years	⊖	28 a	44 a	37 a	23 a	3182 a	53 a	41 a
Static Reach Resources	years	⊖	267 a	349 a	299 a	152 a	6250 a	422 a	73 a
EoL-Recycling Rate [3]	%	⊖	15%	<1%	43-53%	<1%	<1%	<5%	30%

Source: Helbig et al. (2016)

Note: ⊕ means high risk, ⊖ means low risk

Appendix 28:Elemental data on risk of supply reduction

Indicator	Dimension	Risk	Cd	Te	Cu	In	Ga	Se	Mo
By-product dependence	%	⊕	100% (Zn)	100% (Cu, Pb)	9%	100% (Zn)	100% (Bauxite)	100% (Cu)	46% (Cu)
Future technology demand	%	⊕	n/av	n/av	15%	289%	581%	11%	n/av
Substitutability	qualitative	⊖	62	62	30	40	62	53	30

Source: Helbig et al. (2016)

Appendix 29: Elemental data on concentration risk

Indicator	Dimension	Risk	Cd	Te	Cu	In	Ga	Se	Mo
Country Concentration	HHI	⊕	1670	3338	1443	3159	3785	2268	2323
Company Concentration	HHI	⊕	rather low	1108	1108	1867	1667	1108	2183

Source: Helbig et al. (2016)

Appendix 30: Elemental data on political risk

Indicator	Dimension	Risk	Cd	Te	Cu	In	Ga	Se	Mo
Political Stability (WGI-PV)	qualitative	⊖	-0,24	0,07	0,27	-0,35	-0,4	1,07	-0,19
Policy Perception (PPI)	qualitative	⊖	43	55	55	43	47	55	47
Regulation Risk (HDI)	qualitative	⊕	0,79	0,73	0,76	0,80	0,71	0,88	0,79

Source: Helbig et al. (2016)

Appendix 31: Normalized elemental supply risk scores in all 11 indicators

Indicator	Cd	Te	Cu	In	Ga	Se	Mo
Static Reach Reserves	88	76	81	91	0	67	78
Static Reach Resources	0	0	0	38	0	0	82
EoL-Recycling Rate	85	99	57	99	99	95	70
By-product dependence	100	100	9	100	100	100	46
Future Technology Demand	6	14	6	58	83	4	26
Substitutability	38	38	70	60	38	47	70
Country Concentration	61	76	58	75	79	68	68
Company Concentration	0	65	65	73	72	65	76
Political Stability	51	49	49	50	58	34	50
Policy Perception	57	45	45	57	53	45	53
Regulation Risk	79	73	76	80	71	88	79

Source: Helbig et al. (2016)

Appendix 32: Weights of indicators based on different weighting methods

Category	Indicator	AHP weighting	Group weighting	Equal weighting
Risk of Supply Reduction	Static Reach Reserves	6,6%	8,3%	9,2%
	Static Reach Resources	4,0%	8,3%	9,2%
	End-of-Life Recycling Rate	9,3%	8,3%	9,2%
Risk of Demand Increase	By-Product Dependence	8,4%	8,3%	9,2%
	Future Technology Demand	11,2%	8,3%	9,2%
	Substitutability	9,7%	8,3%	9,2%
Concentration Risk	Country Concentration	21,9%	12,5%	9,2%
	Company Concentration	9,4%	12,5%	9,2%
Policy Risk	Political Stability	7,8%	8,3%	9,2%
	Policy Perception	5,5%	8,3%	9,2%
	Regulation	6,1%	8,3%	9,2%

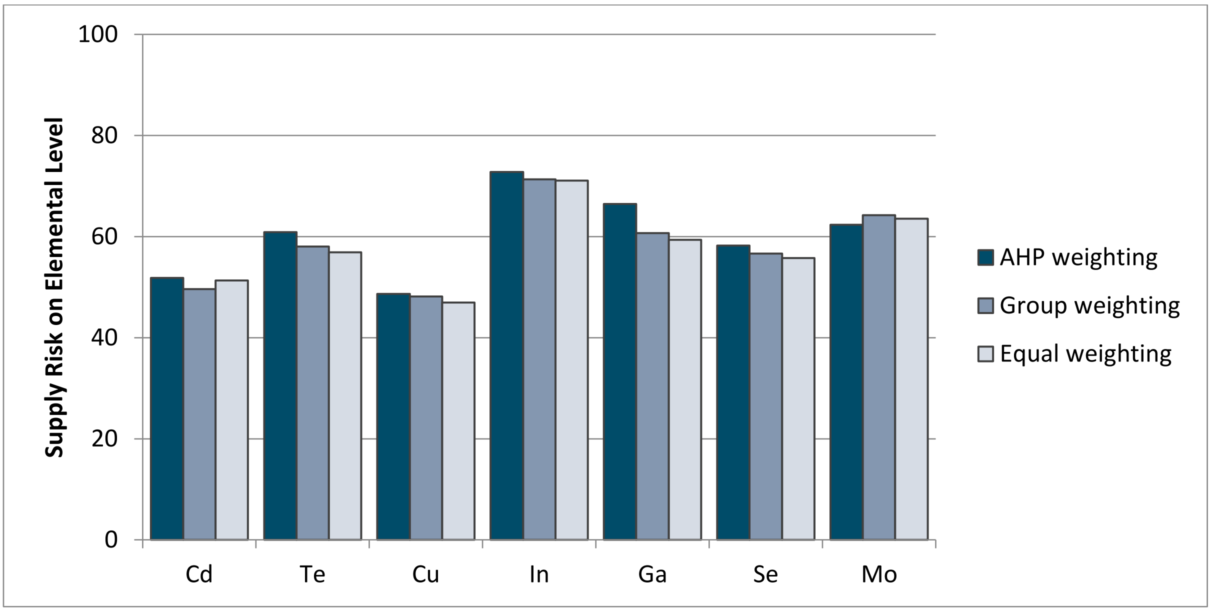
Source: Helbig et al. (2016)

Appendix 33:AHP weighing for supply risk category indicators

Category	Indicator	Weighting
Risk of Supply Reduction (20.0%)	Static Reach Reserves	6,6%
	Static Reach Resources	4,0%
	End-of-Life Recycling Rate	9,3%
Risk of Demand Increase (23.4%)	By-Product Dependence	8,4%
	Future Technology Demand	11,2%
	Substitutability 9.7	9,7%
Concentration Risk (31.3%)	Country Concentration 21.9	21,9%
	Company Concentration 9.4	9,4%
Policy Risk (19.4%)	Political Stability 7.8	7,8%
	Policy Perception 5.5	5,5%
	Regulation 6.1	6,1%

Source: Helbig et al. (2016)

Appendix 34:Overall scores on elemental levels using different scoring methods



Source: Helbig et al. (2016)

Appendix 35: Mass (kg/MW), mass share, cost per kg and cost share of materials in CdTe cells

Data	Cd	Te
Mass, kg/MW	153,4	137,7
Mass share, %	52,7	47,3
Specific material costs, USD/kg	0,86	77,5
Raw material costs, USD/MW	131,92	10672
Cost share, %	1,2	98,8

Source: Helbig et al. (2016)

Appendix 36: Mass (kg/MW), mass share, cost per kg and cost share of materials in CIGS cells

Data	Cu	In	Ga	Se	Mo
Mass, kg/MW	21,0	19,0	2,3	9,6	90,0
Mass share, %	14,8	13,4	1,6	6,7	63,4
Specific material costs, USD/kg	5,089	315,63	144,4	24,11	12,86
Raw material costs, USD/MW	106,97	5993,8	337,9	230,49	1157,4
Cost share, %	1,4	76,6	4,3	2,9	14,8

Source: Helbig et al. (2016)